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Proceedings of a Research Symposium Cosponsored
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PRETREATMENT AND ULTIMATE DISPOSAL OF WASTEWATER SOLIDS

Greetings to Conference
Charles E. Hess
Dean of Cook College, Rutgers University

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Greetings to Conference

Pretreatment and Ultimate Disposal of Wastewater Solids

Dean Charles E. Hess
Cook College, Rutgers University

It is a pleasure to welcome you to the Cook College campus of Rutgers University, It is very appropriate that you meet on the campus of a College dedicated to the concern of man and his environment. As many of you know we had an earlier conference, jointly sponsored by the Environmental Protection Agency and the College, dealing with the land disposal of sewage sludge. It is important to follow that conference with an exploration and evaluation of alternate methods of pretreating and disposing of waste water solids.

We value the close association between the College and the Environmental Protection Agency. Working together on a wide range of environmental problems we hope through research and extension programs to provide answers to many of the serious environmental problems facing the nation's most densely populated state.

I hope that you enjoy your visit to Rutgers University and if we can do anything to make your stay more enjoyable or rewarding, please let us know. We are here to serve you.

OVERVIEW OF SLUDGE HANDLING AND DISPOSAL *

by

J. B. Farrell **

For about 8 years, the Advanced Waste Treatment Research Laboratory, which is now part of U. S. Environmental Protection Agency's National Environmental Research Center, Cincinnati, has had an Ultimate Disposal Section. A primary objective of this group has been to advance the technology for processing and disposing of the concentrates from municipal and advanced wastewater treatment processes. It has thus been our privilege to observe and to take part in the effort to improve the methods for handling and disposal of wastewater sludges.

As the quality of wastewater treatment has improved, sludge handling and disposal have become a greater problem. The trend is not expected to change. As more and more municipalities upgrade facilities to improve effluent quality, the quantity of sludge will continue to increase. Table 1 compares sludge production in 1972 with the estimated production in 1985: the amount of secondary treatment sludge will be almost doubled, and chemical sludges will be produced in quantity; dewatering costs will increase more than proportionately because the biological secondary sludges and several important types of chemical sludges are unusually difficult to dewater.

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* Presented at Research Symposium on Pretreatment and Ultimate Disposal of Wastewater Solids, Rutgers University, New Brunswick, N.J., May 21-22, 1974.

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The operations carried out on wastewater sludge are conveniently divided into treatment operations, such as pumping, thickening, stabilization, and dewatering; and disposal operations, which include incineration, landfill, and landspreading. The more important recent developments in treatment are considered here first.

TREATMENT OPERATIONS

Thickening

If sludges removed from primary and secondary clarifiers are to be dewatered, their solids content should be as high as possible, that is, just short of the point where the "thick" sludge interferes with the operation of the device used for dewatering. Thickening is often required. The most commonly used devices are either gravity or air flotation thickeners. Air flotation thickeners are generally more expensive to operate, but have advantages in certain cases. There is less opportunity for sludge to become anaerobic, because residence time is less than in a gravity thickener and air bubbles that have floated the sludge provide a reservoir of available oxygen. In secondary plants where the effluent has nitrified, gravity thickening of waste activated sludge becomes difficult because the respiring activated sludge organisms use nitrate as their oxygen source after dissolved oxygen is consumed. Nitrogen gas is released and this causes a portion of the sludge to float. If air flotation is utilized instead of gravity thickening, any tendency of sludge to float will, of course, be advantageous.

Stabilization

The conventional stabilization processes are anaerobic and aerobic digestion. The effect of these processes is to reduce odor, reduce the putrefaction potential of the sludge, and reduce the concentration of hazardous microbiological organisms. It is interesting to separate these positive results and consider other means by which they can be carried out. Table 2 measures a number of processes against these characteristics. It is evident that none of the processes do all things well. Further study of Table 2 indicates that certain combinations of processes would be effective, for example, anaerobic digestion followed by pasteurization. These two processes have been carried out in sequence in Germany (1) to treat sludge used on pasture lands during the summer months. Radiation has recently been used following anaerobic digestion (2), again in Germany and for the same purpose.

Costs of combined treatment are additive. Recently, however, there has been interest in the use of anaerobic or aerobic thermophilic digestion. These processes can be operated at 60° C and destroy pathogenic microorganisms. They effectively combine pasteurization and stabilization into a single process, and economies can result.

Most stabilization processes adversely affect supernatants and filtrates from subsequent dewatering operations: anaerobic digestion and heat treatment produce supernatants rich in nutrients and BOD; supernatant from heavy chlorination is high in chloramines and possibly chloro-organic compounds; and supernatant from lime treatment is high in pH. Cost of processing supernatants must be considered in the total cost of processing.

Processes such as digestion and heat treatment reduce the mass of sludge to be treated in subsequent operations. These changes in mass are a cause of great confusion in the literature when costs of complete processing sequences are compared. Costs are most often presented as dollars per ton of dry solids without specifying whether the basis is the sludge mass before or after processing. An unequivocal way of comparing costs of alternative processing sequences for a given wastewater is to base costs on equal wastewater flow (e.g., ¢/1000 gal. of wastewater).

Dewatering

Filtration--The continuous rotary vacuum filter is the most commonly used device for dewatering wastewater sludges. Three different types are used: the drum (cloth on drum, cake removed by doctor blade), cloth belt (cloth belt winds off drum, sharp bend causes cake to drop off, both sides of belt are washed, and cloth returns to drum), and coil (filter surface comprises two layers of tightly coiled springs, otherwise similar to belt type).

A new type of drum filter, utilizing top-feed, has been developed by the Rexnord Corporation for the City of Milwaukee under an EPA grant (see Figure 1). Gravity assists in depositing coagulated solids, which are denser than water, onto the drum surface. Gravity also assists in removing cake from the filter surface, so air blowback to loosen cake is not needed, and the doctor blade is nearly superfluous.

Centrifuges--Solid-bowl continuous conveyor-type centrifuges are the most popular centrifugal device used for sludge dewatering. Cake solids contents are similar to those obtained with vacuum filters. These devices are also excellent classifiers. They are being used with tertiary (3) and primary lime sludges (4) to classify organic and most mineral solids except calcium carbonate into the centrate. The cake is then high in calcium carbonate, which can be calcined and recovered as CaO.

Basket-type centrifuges are being used to thicken aerobically digested sludge. This type of centrifuge will be used to remove solids from the centrate from Los Angeles County's solid-bowl conveyor-type centrifuges (5).

Other Methods--Filter presses are receiving considerable interest in the United States, particularly now that high cake solids is becoming important. At Cedar Rapids, Iowa (6), a filter press has been installed that uses a precoat of sludge incinerator ash and a body feed of incinerator ash, lime, and ferric chloride. Vertical presses, used at several plants in Japan, offer very speedy cake discharge. Precoat is not needed because a section of the filter cloth is washed during each cycle. The cloth advances one frame at each discharge cycle. Thus, a section of cloth is washed after having performed a number of filtrations equal to the number of frames. $Ca(OH)_2$ and $FeCl_3$ are the usual conditioning agents with this filter at Japanese plants.

Belt filters have been used extensively in Europe and are being marketed in the United States. They combine gravity drainage with mechanical pressure after a cake has been formed. The Carter* belt filter, presented schematically in Figure 2, illustrates the principle. The capillary suction filter developed by Westinghouse with the financial support of EPA combines capillary suction dewatering with mechanical pressure. A simplified sketch of the original research unit is presented in Figure 3. Sludge is placed on a dry belt of a foamed porous material, which removes moisture from the sludge by capillary action. A contact roll rides directly on the partially dewatered sludge layer and applies mechanical pressure. Water is forced from the sludge into the porous belt. The sludge cake transfers to the smooth-surfaced contact roll where it is collected and removed. The porous belt, now free of sludge, is washed and squeezed dry. Activated sludges, which can seldom be dewatered to over 15 percent solids, can be dewatered to over 18 percent solids at high rates with this device.

Conditioning

Very few sludges can be filtered without additives of some kind. Lime and ferric chloride continue to be used, particularly when odors are a problem or when improperly digested sludge is to be landfilled. The use of organic polymeric conditioning agents continues to grow.

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* Mention of manufacturer's names does not constitute EPA endorsement.

The most rapid gains, however, have been made by heat conditioning. Sludge heated to 200° C (392° F) for under an hour, in the presence of air or without air, loses its gelatinous nature and becomes easily filterable, generally without chemicals. A substantial portion of the sludge, however, is solubilized and the filtrate must be recycled to biological treatment.

Heat treatment is a satisfactory means for conditioning sludge. Whether or not it is the most cost-effective depends on the situation. The process requires a higher degree of operating skill than is ordinarily available at wastewater treatment plants. It causes odors that are sometimes extraordinarily difficult to eliminate. The additional cost of biologically treating the filtrate (and handling the increased biological sludge generated) should be charged to the process. Even for plants with temporary excess capacity, the loss in capacity caused by the need to process recycle streams must eventually be borne by the wastewater treatment plant. In Great Britain, where effluent quality requirements are more stringent than those in the United States, separate treatment of the supernatant is needed. At one facility, supernatant is being biologically treated by separate activated sludge treatment, which is followed by activated carbon to remove residual COD. At another plant, the entire supernatant will be concentrated by flash evaporation and the concentrate then incinerated.

DISPOSAL OPERATIONS

Hazardous Substances

Disposal of sludge is turning out to be the pivotal question in wastewater processing. Sludges contain the concentrated wastes of a community. It is reasonable to expect that objectionable materials may be present in sufficient concentration to be hazardous. The degree of hazard will, of course, depend on the intended means of disposal.

Some typical concentrations of hazardous substances found in sludge are presented in Table 3. The hazardous substances are toxic metals, toxic organic chemicals, and pathogenic microorganisms. The concentration and quantities of these contaminants clearly limit disposal options. For example, the high concentrations of pathogens in raw sludge virtually prohibit disposal to landfill or to agricultural land. Sludge with a high PCB concentration should not be applied to land if there is a possibility of leaching. Sludge with high mercury content may be suitable for disposal to a landfill but not for incineration. There appears to be no foolproof disposal method suitable for all sludges.

The type of disposal chosen by a community is a commitment not easily changed. Increasing knowledge of hazards, or a change in the nature of the community's wastes, may indicate that the wastewater sludges contain too high a concentration of certain materials for the method of disposal practiced. Two things are necessary--the community should have a current knowledge of the concentrations of hazardous materials in its sludge, and it should have ordinances which allow it to prohibit disposal into its collection systems not only wastes which affect the quality of wastewater processing but also hazardous wastes which are not "neutralized" by the disposal method and are a threat to the environment.

The conventional disposal methods are ocean disposal, landspreading, landfill, and incineration. A discussion of ocean disposal is beyond the scope of this presentation. It appears at this time, however, that disposal of sludge solids by outfalls or by dumping will diminish in coming years, and that the level of hazardous materials in their sludges will have to be reduced by communities continuing to use these procedures.

Land Application

The use of stabilized sludge to fertilize agricultural land or reclaim marginal land is a conserving use which has been practiced for many years. Chicago plans to dispose of a major portion of their sludge by this means (7). Coastal cities are considering it as an alternative to ocean disposal. Attention has been called to possible hazard to crops and to human health from a gradual buildup of toxic metals in soil to which sludge has been applied over a number of years (8). The U. S. EPA is currently studying suggested guidelines for land application of sludge.

It is likely that some restrictions will be placed on the practice of land spreading. Suggestions have been made to limit both the concentrations of metals such as Cd, Cu, Zn, and Ni in the sludge and the maximum loading of sludges on the land (8), or the maximum loading of the metals on the land (9). Knowledge of effects of metals in sludge on crops is sparse, which makes preparation of reasonable guidelines difficult. It is expected, however, that for communities with an unusually high industrial component in their wastewater, recommended application rates may be too low for this method to be competitive with alternative disposal means.

Landfill

Disposition to landfill is the most common way to get rid of sludge. Sometimes the landfill is a properly operated sanitary landfill for disposal of solid wastes and sludge. Often the landfill is an uncovered dumping site inside the plant grounds. Small plants often dispose of their sludge to unprotected sites outside their plant limits--sometimes in flood plains and often without cover.

Little attention has been paid to the disposal techniques practiced by small plants. Disposal to sanitary landfills may be impractical because of distance or because sludges are banned from the landfill, and operators often resort to what is essentially uncontrolled dumping. The EPA is sponsoring work by the U. S. Department of Agriculture to develop a method for operating a private sludge landfill in an ecologically satisfactory way with the use of simple farm machinery.

A properly operated sanitary landfill is an excellent place to dispose a sludge high in metals or persistent organic compounds. The site should be located where groundwater contamination is not possible, and any leachate should be collected and treated.

Incineration

When properly carried out, incineration is a satisfactory means of disposing of the great majority of hazardous sludges. Particulates must be contained by modern scrubbing equipment, and temperature-time characteristics must be adequate to decompose thermally stable organic compounds. Sludges containing mercury are an exception because mercury vaporizes upon incineration and is not captured satisfactorily by conventional scrubbers. Even mercury can be captured, however, if the flue gases are brought to room temperature and filtered (10). A major problem with incineration is poor operation; this can be corrected by good operating procedures and modern control devices.

Trends in Disposal

There has probably not been a more difficult time for forecasting disposal trends. The picture is very negative for ocean disposal, although changes are possible. Land application will be subjected to guidelines--guidelines that will recommend reduced application rates and require more land; landfill is satisfactory, but suitable sites are dimishing rapidly; incineration would appear to face a promising future except that the high cost of fuel, added to the cost of air pollution control equipment, has escalated the total cost.

Promising areas for the future, which are now being seriously investigated, are co-incineration and co-pyrolysis of sludge with solid waste. These methods will not require supplemental fuel. Co-pyrolysis may produce usable fuel gas and char. When estimating future disposal trends (Table 1), incineration and pyrolysis increase from 25 percent in 1972 to 35 percent in 1985. Much of this gain will be from new methods of co-incineration and co-pyrolysis.

Certainly disposal costs, as a proportion of wastewater treatment costs, will rise. The benefit will be a more secure environment for now and for the future.

END

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TABLE 1

TRENDS IN PRODUCTION AND DISPOSAL OF MUNICIPAL WASTEWATER SLUDGE

	1972		1985	
SLUDGE TYPE	Popul. (mill.)	Dry tons** per year	Popul. (mill.)	Dry tons per year
Primary (0.12 lb/cap-da)*	145	3,170,000	170	3,720,000
Secondary (0.08 lb/cap-da)	101	1,480,000	170	2,480,000
Chemical (0.05 lb/cap-da)	10	91,000	50	455,000
	Percent		Percent	
DISPOSAL METHODS				
Landfill	40		40	
Utilized on land	20		25	
Incineration	25		35	
Ocean (dumping and outfalls)	15		0	

* lb x 0.454 = kg

** ton x 0.908 = metric ton

TABLE 2

ATTENUATION EFFECT OF WELL-CONDUCTED TREATMENT PROCESSES ON STABILIZING WASTEWATER TREATMENT SLUDGES

PROCESS	DEGREE OF ATTENUATION		
	Pathogens	Putrefaction Potential	Odor
Anaerobic digestion	fair	good	good
Aerobic digestion	fair	good	good
Heavy chlorination	good	fair	good
Lime treatment	good	fair	good
Pasteurization (70° C)	excellent	poor	poor
Radiation	excellent	poor	fair
Heat treatment (195° C)	excellent	poor*	poor*

* good for filter cake

TABLE 3

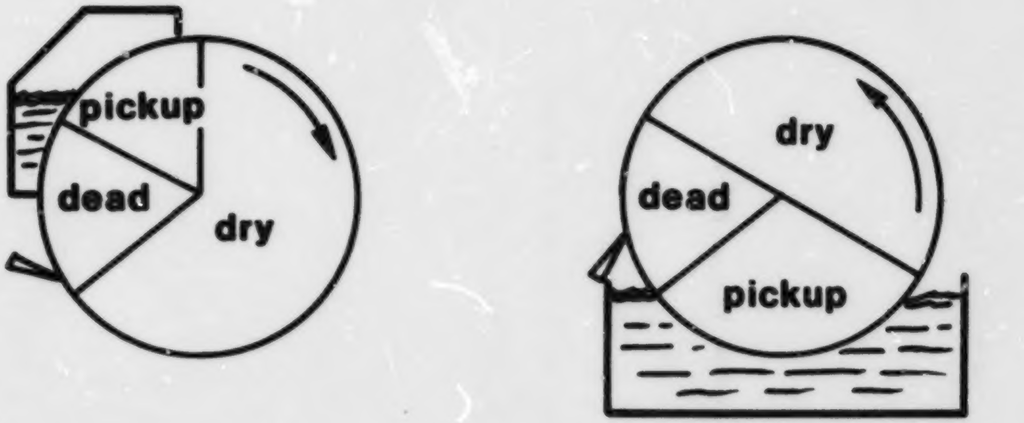
APPROXIMATE CONCENTRATION LEVELS OF SUBSTANCES IN MUNICIPAL WASTEWATER SLUDGES

Sludge parameter	Raw primary	Digested primary
% Volatile solids	72.0	52.6
% Ash	28.0	47.4
Metal concentration (mg/kg dry solid)		
Cd		30
Zn		1,950
Cu		1,000
Ni		350
Hg		5
Bacterial content (per 100 ml liquid sludge)		
Fecal coliform	11 x 10^6	0.4 x 10^6
Salmonella	460	29
Pseudomonas aeruginosa	46,000	34
Organic compounds (mg/kg dry solids)		
Polychlorinated biphenyls		n.d.* to 105
Chlordane		3 to 30
DDT		n.d. to 1
Dieldrin		0.1 to 2.0

* n.d. = not detectable

Figure 1. Drum-Type Vacuum Filters

"Overview of Sluige Handling and Disposal"
by
J. B. Farrell


pickup
dead
dry
dry
dead
pickup

TOP FEED BOTTOM FEED

DRUM-TYPE VACUUM FILTERS

Figure 2. Schematic of Carter Belt-Filter Press

"Overview of Sludge Handling and Disposal"
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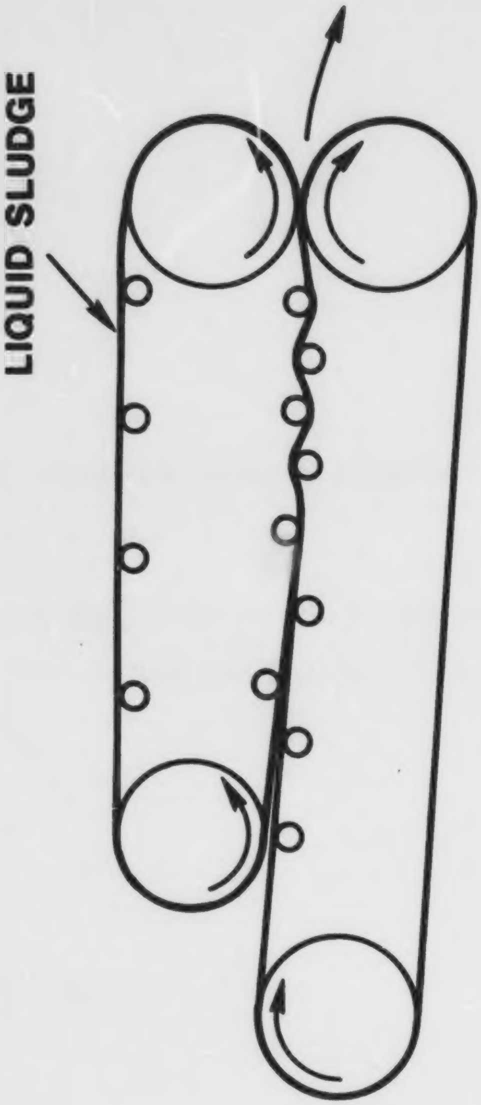

LIQUID SLUDGE

SCHEMATIC OF CARTER BELT-FILTER PRESS

Figure 3. Schematic of Research Capillary Dewatering Unit

"Overview of Sludge Handling and Disposal"
by
J. B. Farrell

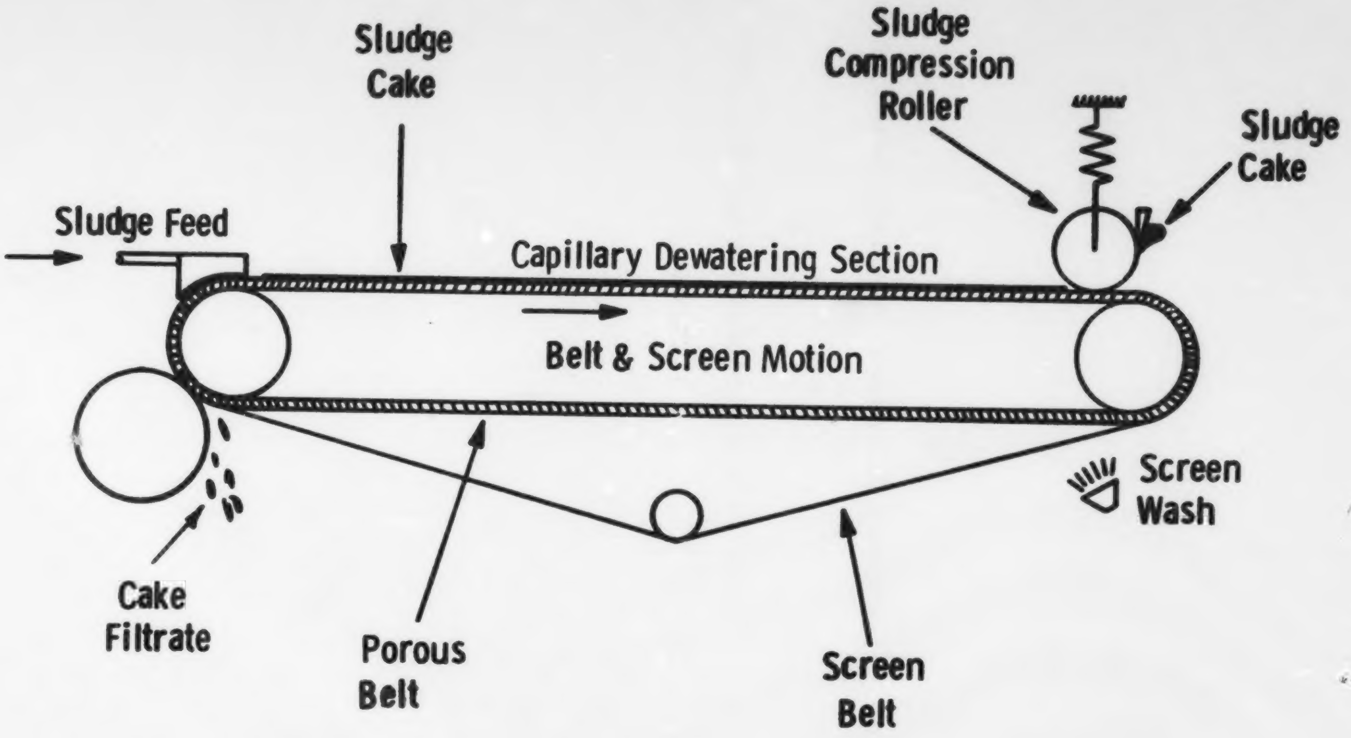
Sludge Cake
Sludge Compression Roller
Sludge Cake
Sludge Feed
Capillary Dewatering Section
Belt & Screen Motion
Screen Wash
Cake Filtrate
Porous Belt
Screen Belt

Figure 3. Schematic of research capillary dewatering unit

ELEMENTAL ANALYSIS OF WASTEWATER SLUDGES FROM 33 WASTEWATER TREATMENT PLANTS IN THE UNITED STATES*

by

B. Vincent Salotto
Ernst Grossman III
Joseph B. Farrell**

Abstract

Sludges, principally of the digested type, from 33 municipal wastewater treatment plants in the United States have been analyzed for 21 metals, Nitrogen, Phosphorus, and Sulfur. The BTU value of some sludges was also determined. Atomic absorption method was used for the determination of metals in sludge. No detectable amount of beryllium was found in any sludge sample analyzed and although very toxic, it evidently does not present a commonly encountered hazard in the disposal of sludge.

Mathematical analysis of the data indicated that the distribution of heavy metals in sludge is approximately log-normal. This behavior is characteristic of all sludge types analyzed thus far. Levels of metals in essentially domestic sludges were found to correspond closely to the 25th percentile distribution of metals in all sludges reported. Comparison of the levels of metals in the United States sludges with corresponding levels in Scandinavian sludges show higher levels in the U. S. sludges. Variation of any particular metal in sludges of a particular wastewater treatment plant was much less than in sludge samples taken from different plants.

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* Presented at Research Symposium on Pretreatment and Ultimate Disposal of Wastewater Solids sponsored by the Environmental Protection Agency and the Department of Environmental Science, Rutgers University, New Brunswick, New Jersey, May 21-22, 1974.

** Research Chemist, Physical Science Technician, and Acting Chief, Ultimate Disposal Section, Treatment Process Development Branch, AWTRL, National Environmental Research Center, EPA, Cincinnati, Ohio 45268.

The data collected are not sufficient to permit analyses of various groupings of data so that the effect of factors such as industrial mix, composition of the water supply, and the type of treatment can be identified. As more data become available, such effects will be investigated. In the near future, major emphasis of continuing work will be to more accurately define the composition of sludges produced by secondary treatment plants processing municipal wastewaters from domestic sources.

INTRODUCTION

The Ultimate Disposal Section of the U. S. Environmental Protection Agency's Advanced Waste Treatment Laboratory, NERC-Cincinnati, has been analyzing wastewater sludges for metals content for the past two and a half years. The major purpose of this effort has been to determine for various sludge types the kinds and levels of metals, especially toxic metals, in sludge. Such interest on the part of EPA is in response to the national interest in the impact of metals on the environment. The treatment and disposal of municipal wastewater sludge, which is the primary concern of the Ultimate Disposal Section, results in a residue which is disposed to land, sea, or soil. One of the disposal means which may be limited by the presence of high concentrations of toxic metals in the sludge is the use of digested sludge as a fertilizer on agricultural land. It is essential to have adequate knowledge of the concentrations of metals found in sludge in order to establish reasonable loadings of sludge on the land.

Initial impetus for the study of hazardous substances in wastewater sludges came as a result of the formation of an EPA task force in the summer of 1971. The task force was organized to determine the effects of sludge incineration on air and land environment. Although emphasis then was on air pollution, data about the sludges to be disposed were still of prime importance. The results of the EPA task force have been published in 1971 which showed wide variation of metal content of sludge (1), and occasionally high levels of pesticides and polychlorinated biphenyls (2).

Municipal wastewater sludges contain a large proportion of cellulose and fecal matter; they also contain a great variety of other undesirable organic and inorganic wastes from household and business activities, including in some cases street runoff. Metal concentrations in wastewater vary from plant to plant, from region to region, and from season to season. Usually, the wastewater received by a wastewater plant represents a mixture of domestic and industrial waste flows. Ordinarily, it is not possible to determine the domestic or industrial component in a wastewater solely on the basis of the metal analysis. For example, copper and zinc come from both domestic and industrial sources. The presence of cadmium does not necessarily indicate an industrial waste because it also may come from household piping and street runoff (from auto tire wear). Similar cases can be cited for other metals. The industrial component in a wastewater is best determined by a detailed investigation of the contributors to the wastewater system.

The fate of metals through wastewater treatment processes depends to a large extent on the chemistry of the metal, the type of biological system employed, and the efficiency of each process step. An investigation of the receipt and fate of metals through wastewater treatment was conducted by this laboratory nine years ago (3). The wastewater treatment plants were surveyed for four metals: chromium, copper, nickel, and zinc. Table 1 shows data taken from that study of incoming and outgoing concentrations of the four metals at each of the wastewater treatment plants. The last column shows for each metal at each plant the percent of the total incoming metal removed by processing and immobilized in the sludges. Removal of any particular metal was not uniform from plant to plant. For example, at Grand Rapids, Michigan, 16 percent of the total copper going through the plant was removed by the sludge, but at the plant at Richmond, Indiana, the sludge removed 73 percent of the copper. These data do not support the generally held belief that the greater part of the metal load going through a wastewater plant will end up in the sludge. Metal removal from a process stream by sludge is an incompletely understood mechanism; evidently many factors, peculiar to any one plant and area served, must be studied before one can predict the fate of metals through the process. Further information on the effects of metals on biological wastewater treatment processes may be obtained from past publications of this laboratory (4).

Recycling sludge to the environment by landspreading is an especially attractive method of disposal because it returns nutrients such as nitrogen, phosphorus, and potassium back to the soil. This method has been in use for centuries in several countries around the world, and is almost an ideal recycling method. There is lately much concern that metals in the sludge may accumulate in the soil to levels where they either are toxic to crops directly, or are taken up by the crops and become hazardous to man or animals that consume them. Webber (5) has documented some of the effects of metals. The group of metals generally regarded as toxic are boron, cadmium, cobalt, chromium, copper, mercury, nickel, lead, and zinc. Cadmium is of great interest now because recent statistical studies indicate that an increase in the body burden of cadmium may increase the incidence of heart disease (6).

The amount of metals a sludge may safely contain and the rate at which sludge may be applied to the land depend on many factors relating to soil condition, climate, crop type, and agricultural practices. Dotson (7) and Chaney (8) have provided guidance and suggestions. Much research must be done before accurate predictions of metal uptake by crops can be made. Despite the incompletely developed state of knowledge, it is evident that the metal content of sludges must be known so that at the least the quantities of metals in the sludges being applied to the land are known.

Generally, wastewater sludge is a brown to blackish, amorphous, heterogeneous, non-Newtonian liquid containing dissolved, colloidal, and suspended solids. Its physical, chemical, and microbiological properties are largely dependent on source of the waste, type of treatment producing the sludge, and sludge-handling steps. Descriptive names for various type sludges refer to the degree of stabilization or treatment. For instance, a digested or digester sludge refers to a sludge stabilized by the anaerobic or aerobic digestion procedure. Unstabilized sludges such as primary or waste activated are referred to as raw sludges. Water content of sludges can range from as little as 55 to 99.9 percent. It contains humus-type organic material, carbon, nitrogen, phosphorus, sulfides, and major constituent metals such as silicon, sodium, potassium, calcium, magnesium, aluminum, and iron. It contains a host of minor or trace metals such as boron, chromium, cobalt, copper, cadmium, lead, manganese, nickel, silver, and zinc. A trace metal can be present in concentrations as high as 11,000 mg/Kg in the sludge on a dry basis. This translates to 1.1 percent and is evidently no longer present in "trace" amount.

COLLECTION AND ANALYSIS OF SLUDGES

From the summer of 1971 to January 1974, about 100 sludge samples have been analyzed for content of metals and other constituents such as volatile solids, nitrogen (TKN), phosphorus, sulfur, and heat of combustion. These samples have been collected from 33 wastewater treatment plants in 13 states of the U. S. (see Fig. 1 and Table 2). They are not in any manner broadly representative of wastewater sludge around the country; nor have they been selected according to a prearranged schedule. The analysis program was commenced in response to the study by the EPA task force of the effects of sludge incineration on land and air (1). Locations were chosen at sites thought to be suitable for incineration tests. Later sampling concentrated on the Eastern Seaboard where the practice of ocean disposal was being questioned. Also a sampling program was originated by the EPA regional staff in California and by the Ultimate Disposal staff in Ohio, where it was convenient for this laboratory group to collect samples.

The treatment plants sampled range in size from small plants in urban communities (about 0.1 mgd) to large plants in the larger cities such as Baltimore and Philadelphia (about 150 mgd). Some plants such as those along the New York-New Jersey coastal tier area employ primary treatment only, disposing or barging the digested sludge to an ocean dumping area. Wastewater treatment plants with secondary aeration facilities produce waste-activated sludges. Most plants employ anaerobic digestion of their sludge so that a great percentage of samples received for analysis were digested sludge.

A variable number of sludge samples were submitted by the 33 treatment plants. Some plants submitted one sample; others such as the Millcreek Plant in Cincinnati, Ohio, submitted 25 samples. To provide for equal statistical treatment of the data from the 33 plants, the results of the analyses of any one plant were averaged and treated as one entry from that plant. Obviously, no averaging was needed in those cases in which only one sample of sludge was submitted.

Routine tests on sludge were performed according to Standard Methods (9). The heat of combustion tests were done using a Parr* bomb apparatus. A pretreatment technique was employed in the analysis of metals in the sludge. This step was necessary in order to convert metal forms to ionic soluble forms capable of determination by the atomic absorption procedure. Analysis proceeded using a weighed and dried sludge sample. A combination of nitric and sulfuric acids was used to completely decompose the sample except for residual silicious material that all sludge samples contain. As a final step in the digestion procedure, a few milliliters of perchloric and nitric acid mixture were used to clear and remove the last traces of charred organic matter. There is nothing new about this pretreatment step except that other laboratories may use other combinations of acids to decompose the sludge sample. A reagent blank was carried through the metals analyses in order to minimize errors due to contamination from glassware and reagents. Metal concentrations in the blank were low, indicating no gross contamination from reagents and glassware used in the procedure.

A method of standard addition was made to determine the percent recoveries of metals. The test gives valuable information as to losses of metals either through volatilization or interferences. This was done by adding known concentrations of metals standards to sludge samples and determining the total metal content of the sludge. Thus, knowing the total and metal content originally determined in the sludge sample, one could calculate the amount of added metal recovered by the procedure. The standards contained cadmium, copper, mercury, manganese, nickel, lead, and zinc, and were

*Mention of products and manufacturers is for identification only and does not imply endorsement by the United States Environmental Protection Agency.

determined in the presence of other metals normally found in sludge. The percent recoveries of metal standards added to a Little Miami Wastewater Treatment Plant sludge sample, Cincinnati, Ohio were as follows:

Metal Added	Original Concentration in the Sludge* mg/Kg	Amount Added mg/Kg	Total Metal Found mg/Kg	Standard Recovery mg/Kg	Percent
Cadmium	150	500	650	500	100
Copper	1500	1000	2500	1000	100
Manganese	300	500	850	550	110
Nickel	400	500	850	450	90
Lead	600	1000	1500	900	90
Zinc	3800	3500	7620	3820	109

* Original analysis of digested sludge sample from Little Miami Wastewater Treatment Plant, Cincinnati, Ohio

As can be seen from the percent column, recoveries were in the range of 90-110 percent, which indicates an accuracy of ±10 percent in results of analyses.

Because of the volatility of mercury and its compounds, sludge samples for determinations of mercury were treated differently from samples used for analyses of the other metals. A weighed and dried sample of the sludge was cold-digested with a solution of sulfuric, nitric acids, and potassium permanganate overnight in a stoppered, Erlenmeyer flask. This treatment decomposes the sludge sample and converts mercury forms to Hg^{2+} without loss of mercury. Subsequent analysis by the vapor atomic absorption procedure (10) determines as little as 0.5 mg/Kg mercury concentration in a dried sludge sample.

The regular atomic absorption procedure for most metals generally does not warrant reporting results less than 100 mg/Kg (ppm). Since cadmium often falls below this concentration, an improved procedure had to be adopted for this important element. Consequently, to determine cadmium content of sludges, a graphite furnace was used in conjunction with the atomic absorption spectrophotometer to report concentrations as low as 10 mg/Kg. For the more common elements in sludge, concentrations were substantially greater than 100 mg/Kg, so the graphite furnace was not needed.

All metal analyses in this paper are reported on a dry basis of sludge because of the variable wet condition of the sludge samples as received by the laboratory. Results are shown as milligram of metal per kilogram of dry sludge. This appears to be a universally acceptable method of reporting as it is identical to reporting on a part per million basis.

Much remains to be done in improving and developing analytical methods for the determination of metals in sludge. The lack of uniform methods or analysis is directly traceable to the complex nature of the sludge medium. Van Loon, et al (11) have written an excellent discussion about the analyses of sludge.

STATISTICAL APPROACH

As can be anticipated from the above discussion, which pointed out the wide variety of plants sampled, concentrations of individual metals varied over wide ranges. It was necessary to investigate the best means for indicating:

a) the central tendency of the data,

b) deviations of the dispersion of data points from the central tendency.

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b) deviations of the dispersion of data points from the central tendency.

The first steps attempted in the analysis of the data utilized a graphical method. The data points were first tabulated in the order of ascending concentrations and a cumulative frequency distribution was calculated for any particular metal. These results were plotted on normal probability paper, which would yield a straight line if the distribution were normal. Both concentration and logarithm of concentration were plotted on normal probability paper. The distribution of the logarithm of concentration was tested, because a logarithmic normal distribution (log-normal) is commonly encountered in analytical work when the concentrations being measured are small (near zero) and the samples are heterogeneous. With the nature of the distribution established, a computer was used to calculate 25th, 50th, 75th, 90th and 95th percentile values, arithmetic mean concentration and standard deviation, and logarithmic mean concentration and its associated standard deviation. The antilog of the logarithmic mean was calculated giving the familiar geometric mean. The computation of the antilog of the standard deviation gave the spread, which is a multiplying and dividing factor for the geometric mean. This is quite different from the standard deviation of an arithmetic distribution which is an added and subtracted factor. Results of analyses reported as less than the detectalbe limit of the analytical method were not included in the calculation of the means and standard deviations of the arithmetic and logarithmic distributions. Generally, the number of samples excluded from the tabulation was about 10 percent of the total. If more than 30 percent of the total number of entries were reported as less than some stated value, the statistical parameters were not calculated for that particular metal. On the other hand, the calculation of the median and the various percentiles for any particular metal did include values above and below a specified limit of analysis. Thus, the comparison of the median with the other means shows the influence of the less-than-detected values. The median is always an important statistic in a distribution because it is unbiased (by definition it is always the midpoint of a distribution). Unfortunately, the measure of dispersion used with the median (such as the range or modifications of the range) are not as usefully related as the standard deviation to the nature of the distribution. This reduces the value of the median as a statistic. Nevertheless, because of its unbiased nature and simplicity the median is generally determined along with the other more sophisticated measures of central tendency.

METALS IN SLUDGE

Distribution of Metals Found in Sludge--In all the tabulations which follow, sludges are shown as two types. Eighty of the 100 samples analyzed were digested sludges and the remainder were the raw or primary unstabilized sludges. Digested sludge is the most important because this is the most common type of stabilized sludge disposed to the land.

Cumulative distributions of the frequency of occurence versus both concentration and logarithm of concentration were plotted for each of the 21 metals. The distribution of logarithm of concentration showed a closer approach to linearity than concentration, indicating that concentration is log-normally distributed. Frequency histograms of the logarithm of concentration are shown in Figures 2 through 7 for 4 metals--cadmium, copper, lead and zinc. As the curves illustrate, the distributions are not skewed when presented on this basis, and approximate the typical bell shape of a normal distribution.

Calculations of statistical indices were made for about 21 metals in digested sludge samples. The three measures of central tendency are presented for 13 metals in Table 3 to illustrate how each statistical approach can give different results. Note that for each particular metal the arithmetic mean is greater than the geometric mean, which in turn is greater than the median. The most bias is shown in the arithmetic mean when compared to the median. This is undoubtedly due to the effect of a few high values. On the whole the geometric mean was much closer than the arithmetic mean to the median for any particular metal, and thus appears to be a better measure of central tendency. This conclusion is supported by the fact that the log-normal distribution fits the data. The geometric mean and the median should then be identical, except for some displacement which results because "less-than-detectable" points have been included in the calculation of the median but not in the calculation of the geometric mean.

The spread associated with the geometric mean was greater for some metals such as cadmium, chromium, and manganese than for others. This is an indication that these metals are not ordinarily found in domestic wastewater but are occasionally introduced (probably from industrial sources) in relatively high concentrations.

The metals concentration data for digested sludges are presented in Table 4 as percentiles. The non-linear nature of the distribution can be seen by comparing the concentration "step" from the 25th to the 50th percentile with the 50th to the 75th. If the distribution were arithmetic, these steps would be equal. In all cases, the 50th to 75th percentile step is the greater, indicating a peak in the distribution at some concentration below the 50th percentile.

The 3 measures of central tendency are shown for metals in raw or unstabilized sludge in Table 5. As was tho case of the averages in digested sludge, the geometric mean and median are better measures of central tendency than is the arithmetic mean.

The geometric mean levels of metals concentration in raw or unstabilized sludge are compared in Table 6 to the concentrations in digested sludge. Concentrations of chromium, copper, lead and zinc were higher in digested sludge than in raw sludge. This is an expected result because organic matter is volatilized during anaerobic digestion leaving metals behind in a more concentrated form. However, concentrations of some other metals were higher in raw sludge than in digested sludge. The most probable cause of this anomaly was the fact that the sludge samples were not from the same plants.

The average concentrations shown thus far are based on analyses of samples from 33 plants. Obviously as more samples from more wastewater plants are examined in our future research, the accuracy of the averages will be improved as the base is broadened.

Toxic and some nontoxic metals have been included in this study to make reporting of metals in sludge as complete as possible. Some metals have not been reported either because the data base was insufficient or because some metals such as iron and aluminum simply do not present any hazard or toxicity to plants and man. In the former class are such metals as arsenic, barium, beryllium, molybdenum, and antimony. Beryllium is of special interest because of its reported high toxicity to man (12). Although the analytical detection limit was low for this metal as compared to other metals (less than 2 mg/Kg), beryllium was not detected in any of the analyzed 102 sludge samples. It appears at this point that beryllium in sludge presents little or no problems in sludge disposal.

Comparison of Data of This Study with Other Sources--Dean and Smith (13) have presented a tabulation of geometric mean metals concentrations which includes information taken from the U. S. literature and from Scandinavian sources (14, 15) as well as from the present investigation. U S. metal levels are higher than the levels in Denmark and Sweden. In Table 7, their tabulation is reproduced but data from this laboratory have been updated to include additional data and results for cadmium. Swedish and Danish national averages of 8 toxic metals tend to agree with each other. Similarly, U. S. literature values and results from the present investigation agree, but are generally at a higher level than the Scandinavian results. Some reasons for this are obvious such as the fact that there is a greater level of industrial activity in the typical community in the United States.

Dean and Smith (13) have shown that both the Scandinavian and U. S. data follow a log-normal distribution. This was also found to be true in this study, and it appears that the log-normal distribution is characteristic of metals concentrations in sludge. The log-normal characteristic is probably attributable to two factors. First, the sampled communities vary widely among themselves. The result is a variation which is large compared to the mean. The distribution is bounded on one side only (because concentrations are never less than zero). This combination of effects generally forces a distribution to be skewed. Second, there are a few dominating influences which are probably log-normally distributed; for example, the proportion of industrial wastes in the wastewater flow. Consequently, skewness of the proportion of industrial wastes in relation to the other sources of the wastewater flow leads to skewness of the distribution of any particular metal in the waste sludge. The good fit of the metals data to a log-normal distribution should be considered an empirical observation with little theoretical justification. Moreover, it has been consistently observed in this study and other investigations (13) and can be used with confidence.

Toxic Metals in Sludge--Chaney has proposed that sludge application to agricultural* land is a reasonable practice provided a) there is a good benefit: risk ratio, and b) metals loadings are limited to permit continued general* farming. He has analyzed a number of sludges and has presented a listing of the reasonably attainable minimum toxic metal content. Higher than minimum quantities are attributed to industrial pollution. He states that these metal contents are attainable and reasonable for digested sludge, and that sludges with higher toxic metal content should not be applied to agricultural land.

Chaney's tabulation is presented below:

Element	Content
Zinc	< 2,000 ppm
Copper	< 800 ppm
Nickel	< 100 ppm
Cadmium	< 0.5% of Zn
Boron	< 100 ppm
Lead	< 1000 ppm
Mercury	< 15 ppm

Of those metals, cadmium is the major threat to human health. Chaney (8) states that the only apparent way to be sure that the cadmium in a food crop grown on a sludge or effluent-treated soil will not be a potential food-chain hazard is to reduce the cadmium content of sludges to 0.5 percent of the zinc content. In this way, zinc excess (at about 500 ppm zinc in leaves) would injure the crop before the zinc or cadmium content of the crop constituted a health hazard.

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* Emphasis is Chaney's

Chaney's recommended concentrations are compared with the data obtained in this study in Table 8. Median (i.e. 50th percentile) and 25th percentile concentrations are shown for the data from the 33 plants as well as median concentrations for domestic sources. The "domestic source" plants were selected from the 33 plants on the basis of location away from industrialized areas and on size of the plant. It is interesting to note that the 25th percentile concentrations and the domestic source concentrations coincide very closely. Empirically, it can be said that the domestic source and the 25th percentile compositions are essentially the same.

The median concentrations obtained in the present investigation exceed Chaney's values for 5 out of 7 metals, but the domestic sources and the 25th percentile values are exceeded by Chaney's for about half of the metals. The absolute magnitude of Chaney's values for any given metal, however, do not match well with the domestic source or the 25th percentile.

There are threefold differences in the values for boron, mercury, and nickel. Median, domestic source, and 25th percentile compositions show cadmium to zinc ratios of 1.12%, 0.79%, and 0.91%, which substantially exceed Chaney's recommended value of 0.5%. There appear then to be substantial differences between Chaney's sludge "free of industrial pollution" and the present study's "domestic source" sludge. The differences probably arise from differences in the geographical area covered and the types of plants selected for the samples.

If the metals content of a sludge is to be compared against a standard composition--consider the median composition--the probability that the concentrations of each metal will be less than or equal to the median will be appreciably less than 50 percent. To illustrate this point, the over 100 sludge sample analyses (from 33 plants) were compared against Chaney's standard composition and against the 25th percentile composition. Only 1 sludge met Chaney's standard composition and 3 sludges met the 25th percentile composition. It is evident that very few sludges will meet even a 25th percentile composition. A 75th percentile composition would exclude about 50 percent of the sludges, and would serve the purpose of excluding sludges with a high proportion of industrial wastes.

One implied problem in the above discussion is the definition of "domestic source." A city can be considered a domestic source even if its wastewater contains the discharges from the small businesses that serve the needs of the city. If there is a substantial industrial discharge, it no longer can be considered a domestic source. A survey of domestic sources should be geographically diverse and should include large as well as small balanced communities. It should not overrepresent wastewater plants serving surburban areas which sometimes receive only household wastewater. When a large number of plants are surveyed without detailed knowledge of each community served, a significant contribution from industrial sources cannot be ruled out. On the other hand, a high concentration of certain metals may indicate an unusual domestic activity and is not prima facie evidence of an industrial discharge.

Variation of Metals in Sludge Samples from Cincinnati's Millcreek Plant--The experiments with digested sludge from the Millcreek Wastewater Treatment Plant were undertaken to establish the extent of variation in metals content of sludge from a given plant. This was done by collecting a digested sludge sample from the treatment plant each day for a period of three weeks. Each sample was analyzed for metals content, and the geometric mean and spread was calculated for 11 metals. These results are shown in Table 9. Recall that the spread is a dividing and multiplying factor. For example, dividing and multiplying the chromium geometric mean value of 3360 mg/Kg by 1.30 gives a spread of approximately 2590-4370 mg/Kg. A spread of 1.00 indicates no deviation at all. The percent of data not detected is shown to indicate that in most cases 90 or more percent of the analytical results were included in the computation of the geometric means.

Geometric mean composition and spread for the Millcreek sludges are compared with the values obtained for the 33 plants (Table 3) in Table 10. Metal concentrations in the Millcreek sludge are higher than for the 33 plants and reflect the industrial activity in the Millcreek Valley region of Cincinnati. The spreads are much lower for the Millcreek data than the spreads for the 33 plants. The Millcreek spreads are a "within-plant" variation, which includes analytical and sampling error as well as actual changes in the sludge. The spreads for the 33 plants represent a "between-plant" variation, which is evidently much larger than the within-plant variation. The results indicate satisfactory precision of the analytical work. A sampling frequency of about once every two weeks ought to be satisfactory to obtain a knowledge of metals content of sludge from any one plant. This same conclusion may not hold true when raw sludge is sampled because the averaging effect obtained when sludge is held in a well-mixed digester for a residence time of two weeks is not present. Because of low variability of within-plant measurements, care must be taken in including them in a survey to establish between-plant variability. In this and in the Scandinavian (14, 15) studies, this difficulty was avoided by averaging analyses at a given plant and using them as a single entry in the data array. One advantage, which should not be overlooked, is that within-plant measurements help to more accurately establish the constituent characteristics of a sludge from any given plant.

Estimating Metal Concentrations in the Influent Stream to a Wastewater Plant--Having arrived at average concentrations of metals in sludge, one can make a rough estimate of the concentrations of the common metals found in wastewater influent streams. A knowledge of the average sludge production is needed for this calculation. Fair and Geyer (16) estimate that an average of 890 pounds of dry digested sludge solids are produced from treating one million gallons of wastewater. This is equivalent to 0.107 gram of digested sludge per liter of raw wastewater. Assume that 70 percent of each of the 8 toxic metals in Table 3 is removed in the sludge. The calculation is illustrated for zinc which has a median concentration of 2.78 mg/Kg (Table 3):

$$\text{Metal removed from wastewater} = \text{Metal found in sludge}$$

$$(1 \text{ liter})\ (0.70)\ (x \text{ mg/l}) = (0.107 \text{ gram})\ (2.78 \text{ mg/g})$$

$$x = 0.424 \text{ mg/l estimated zinc concentration in raw wastewater}$$

Calculated concentrations of other metals based on median concentrations in the sludge shown in Table 3 are as follows:

Metal	Concentration mg/l
Cadmium	0.005
Chromium	0.170
Copper	0.190
Mercury	0.001
Manganese	0.060
Nickel	0.060
Lead	0.130
Zinc	0.420

The calculated values show that, in order to detect metals such as cadmium, mercury, manganese, and nickel in raw wastewater, very sensitive analytical procedures must be used because concentrations of these metals are very low. This calculation procedure may be reversed, and used to estimate metal content of sludge. The approximate nature of the calculation should be recognized.

ADDITIONAL SLUDGE PARAMETERS

In addition to the metal analysis, nitrogen, phosphorus, sulfur, volatile solids, and heat of combustion were determined on sludge samples. An oxygen bomb calorimeter procedure was used to determine the heat of combustion of sludge (17). Geometric mean values of these constituents are shown in Table 11. The average heat of combustion of raw primary sludge, as expected, was higher than that of digested sludge, due to the higher percentage of volatile organic matter in raw primary sludge. Phosphorus and sulfur, on the other hand, are concentrated in the digested sludge, since these constituents are only partially lost during anaerobic digestion. Nitrogen concentration is lower in digested sludge than in raw primary sludge, probably due to the conversion of nitrogen forms to soluble or volatile forms of nitrogen such as NH_4-N and NH_3-N.

CONCLUSIONS

The following conclusions can be drawn from this investigation:

1. In metal analysis of sludges, the distribution of the concentration values of any one particular metal is characteristically log-normal.
2. Concentrations of toxic metals in sludges from 33 wastewater plants in the United States are higher than those found in the sludges of two Scandinavian countries.
3. Levels of metals found in domestic sludge are higher than those recommended to be added to agricultural soil.
4. Within-plant variation of any one metal in sludge samples from a single plant is much less than the variation of the same metal in between-plant sludges.

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TABLE 1

FATE OF METALS THROUGH SEWAGE TREATMENT PROCESS(a)

Bryan, Ohio, Activated Sludge

Metal	Influent Sewage, (mg/l)		Final Effluent, (mg/l)		Percent Immobilized in Sludge
	Average	Range	Average	Range	
Chromium	0.8	0.6-1.1	0.2	0.2-0.3	-
Copper	0.2	0.2-0.3	0.1	0.04-0.1	-
Nickel	0.05	0.03-0.1	0.05	0.03-0.1	-
Zinc	2.2	1.4-3.0	0.2	0.2-0.3	-
Grand Rapids, Michigan, Activated Sludge					
Chromium	3.6	0.7-5.6	2.5	1.0-3.3	40
Copper	1.4	0.7-2.4	1.6	0.4-2.9	16
Nickel	2.0	1.3-3.4	1.8	1.0-2.5	12
Zinc	1.5	0.6-2.5	0.8	0.6-1.2	58
Richmond, Indiana, Activated Sludge					
Chromium	0.8	0.2-2.1	0.2	0.01-0.5	82
Copper	0.2	0.1-0.4	0.07	0.04-0.2	73
Nickel	0.03	0.01-0.1	0.02	0.01-0.03	78
Zinc	0.3	0.1-0.5	0.1	0.1-0.2	85
Rockford, Illinois, High-Rate Trickling Filter					
Chromium	1.8	0.5-2.9	1.2	0.6-1.5	37
Copper	1.4	0.6-3.3	1.0	0.5-3.6	23
Nickel	0.9	0.2-1.9	0.9	0.5-1.4	8
Zinc	2.7	1.2-3.4	1.3	0.8-1.7	53

(a) Data taken from report of E. F. Barth, et al (3).

TABLE 2

WASTEWATER TREATMENT PLANT LOCATIONS

- ARIZONA
  - PHOENIX
  - TUCSON
- CALIFORNIA
  - BARSTOW
  - CARSON
  - FRESNO
  - LOS ANGELES
  - MISSION BAY
  - MONTEREY
  - OXNARD
  - SAN FRANCISCO
  - SAN LORENZO
  - SAN MATEO
  - TAHOE
- HAWAII
  - PEARL CITY
- INDIANA
  - INDIANAPOLIS
- IOWA
  - CEDAR RAPIDS
- MARYLAND
  - BALTIMORE
- NEVADA
  - LAS VEGAS
  - RENO
- NEW JERSEY
  - BERGEN COUNTY
  - ESSEX COUNTY
  - MIDDLEBORO
  - N.W. BERGEN COUNTY
  - PASSAIC VALLEY
- NORTH CAROLINA
  - GREENSBORO
- OHIO
  - CINCINNATI
  - COLUMBUS
  - DAYTON
- PENNSYLVANIA
  - NORRISTOWN
  - PHILADELPHIA
- VIRGINIA
  - LORTON
- WASHINGTON
  - EDMONDS
  - LYNNWOOD

TABLE 3

AVERAGE CONCENTRATIONS OF METALS IN DIGESTED SLUDGE

(ALL FIGURES MG/KG DRY SLUDGE BASIS)

METAL	ARITHMETIC MEAN	ARITHMETIC STD.DEV. (+ and -)	GEOMETRIC MEAN	GEOMETRIC STD.DEV.(a) (÷ and X)	MEDIAN 50% VALUE
SILVER	250	230	190	1.99	100
BORON	430	310	380	1.58	350
CADMIUM	75	104	43	2.47	31
CALCIUM	36,500	23,800	31,100	1.77	30,000
CHROMIUM	1,860	1,920	1,050	3.22	1,100
COBALT	350	220	290	1.88	<100(b)
COPPER	1,590	1,670	1,270	1.95	1,230
MERCURY	10	18	6.5	2.34	6.6
MANGANESE	1,300	2,290	475	3.67	380
NICKEL	680	620	530	1.88	410
LEAD	2,750	2,350	2,210	1.82	830
STRONTIUM	520	670	290	2.70	175
ZINC	4,210	3,800	2,900	2.40	2,780

(a) Standard deviation of the geometric mean is a ratio and has no units.

(b) The symbol "<" indicates less than.

TABLE 4

QUARTILE DISTRIBUTION OF METALS IN DIGESTED SLUDGE

(ALL FIGURES MG/KG DRY SLUDGE BASIS)

METAL	PERCENTILE GROUPING PERCENT EQUAL TO OR LESS THAN STATED VALUE					NO. OF TREATMENT PLANTS
	25%	50%	75%	90%	95%	
SILVER	<100	100	230	430	450	23
BORON	300	350	450	490	870	13
CADMIUM	15	31	67	80	140	18
CALCIUM	22,800	30,000	45,200	49,000	58,800	23
CHROMIUM	395	1,100	2,700	4,000	5,820	23
COBALT	<100	<100	260	500	520	23
COPPER	900	1,230	2,120	2,970	4,310	24
MERCURY	5.1	6.6	8.5	11	12	22
MANGANESE	160	380	770	4,860	6,080	24
NICKEL	300	410	550	1,170	1,720	24
LEAD	<500	830	2,000	2,770	4,370	24
STRONTIUM	100	175	340	1,490	1,790	22
ZINC	1,640	2,780	5,000	9,880	11,100	24

TABLE 5

AVERAGE CONCENTRATIONS OF METALS IN RAW SLUDGE

(ALL FIGURES MG/KG DRY SLUDGE BASIS)

METAL	ARITHIMETIC		GEOMETRIC		MEDIAN 50% VALUE
	MEAN	STD.DEV. (+ and -)	MEAN	STD.DEV.(a) (÷ and X)	
SILVER	490	370	355	2.51	<100
BORON	880	410	775	1.67	805
CADMIUM	30	15	27	1.53	20
CALCIUM	13,800	7,830	11,700	1.82	13,900
CHROMIUM	1,370	1,400	940	2.75	750
COBALT	700	770	410	2.32	240
COPPER	860	550	740	1.67	660
MERCURY	15	23	8.2	2.54	5.5
MANGANESE	1,310	2,860	460	3.32	200
NICKEL	580	540	420	2.12	335
LEAD	1,380	775	1,150	1.95	1,150
STRONTIUM	190	75	175	1.45	<100
ZINC	1,960	1,000	1,740	1.66	1,880

(a) Standard deviation of the geometric mean is a ratio and has no units.

TABLE 6

COMPARISON OF METALS IN RAW TO METALS IN DIGESTED SLUDGES

(ALL FIGURES MG/KG DRY SLUDGE BASIS)

METAL	RAW SLUDGE GEOMETRIC MEAN	DIGESTED SLUDGE GEOMETRIC MEAN
SILVER	355	190
BORON	775	380
CADMIUM	27	43
CHROMIUM	940	1,050
COBALT	410	290
COPPER	740	1,270
MERCURY	8.2	6.5
MANGANESE	460	475
NICKEL	420	530
LEAD	1,150	2,210
STRONTIUM	175	290
ZINC	1,740	2,900

TABLE 7

COMPARISON OF METALS IN UNITED STATES SLUDGES WITH METALS IN SCANDINAVIAN SLUDGES

(ALL FIGURES MG/KG DRY SLUDGE BASIS)

METAL	THIS STUDY	U. S. SOURCES[(a)]	DANISH DATA[(b)]	SWEDISH DATA[(c)]
	GEOMETRIC MEAN CONCENTRATIONS			
CADMIUM	43	69	10	9.8
CHROMIUM	1,050	840	110	170
COPPER	1,270	960	340	670
MERCURY	6.5	28	7.8	5.8
MANGANESE	475	400	350	400
NICKEL	530	240	37	65
LEAD	2,210	480	470	220
ZINC	2,900	2,600	2,600	1,900

(a) SEE REFERENCE 13

(b) SEE REFERENCE 14

(c) SEE REFERENCE 15

TABLE 8

TOXIC METALS CONTENT OF DOMESTIC SLUDGE

(ALL FIGURES MG/KG DRY SLUDGE BASIS)

TOXIC METAL	THIS STUDY AVERAGE 33 PLANTS(a) MEDIAN	DOMESTIC SOURCE(b) MEDIAN	25TH PERCENTILE(c)	CHANEY'S DATA(d)
BORON	350	300	300	100
CADMIUM	31	14	15	<10
CHROMIUM	1,100	300	395	--
COPPER	1,230	700	900	<800
MERCURY	6.6	6.8	5.1	<15
MANGANESE	380	100	160	--
NICKEL	410	300	300	<100
LEAD	830	<500	<500	<1,000
ZINC	2,780	1,700	1,640	<2,000

(a) SEE DATA OF TABLE 3

(b) AVERAGE BASED ON ANALYSES OF SLUDGES FROM PLANTS RECEIVING DOMESTIC WASTES

(c) SEE DATA OF TABLE 4

(d) SEE REFERENCE (8)

TABLE 9

AVERAGE CONCENTRATION AND SPREAD OF METALS IN SLUDGE SAMPLES FROM THE MILLCREEK PLANT IN CINCINNATI, OHIO

METAL	PERCENT NOT DETECTED	GEOMETRIC MEAN MG/KG	SPREAD (÷ and X)
SILVER	5	190	1.24
CADMIUM	10	130	1.51
CHROMIUM	0	3,360	1.30
COBALT	5	515	1.07
COPPER	0	3,550	1.31
MERCURY	0	1.8	1.60
MANGANESE	0	2,360	1.59
NICKEL	5	1,590	1.17
LEAD	0	6,980	1.49
STRONTIUM	0	2,220	1.07
ZINC	0	10,700	1.22

TABLE 10

COMPARISON OF VARIATIONS OF METALS IN SLUDGES
ONE PLANT VS. MANY PLANTS

(MEAN EXPRESSED IN UNITS OF MG/KG)

METAL	MILLCREEK PLANT CINCINNATI, OHIO MEAN X SPREAD(a)	AVERAGE 33 PLANTS MEAN X SPREAD(a)
SILVER	190 x 1.24	190 x 1.99
CADMIUM	130 x 1.51	43 x 2.47
CHROMIUM	3,360 x 1.30	1,050 x 3.22
COBALT	515 x 1.07	290 x 1.88
COPPER	3,550 x 1.31	1,270 x 1.95
MERCURY	1.8 x 1.60	6.5 x 2.34
MANGANESE	2,360 x 1.57	475 x 3.67
NICKEL	1,590 x 1.17	530 x 1.88
LEAD	6,980 x 1.49	2,210 x 1.82
STRONTIUM	2,220 x 1.07	290 x 2.70
ZINC	10,700 x 1.22	2,900 x 2.40

(a)The spread is here expressed as the antilog of the geometric standard deviation.

TABLE 11

AVERAGE CONCENTRATIONS OF OTHER CONSTITUENTS IN PRIMARY AND DIGESTED SLUDGE

(MG/KG EXCEPT WHERE NOTED)

CONSTITUENT	PRIMARY SLUDGE GEO. MEAN X SPREAD	PRIMARY SLUDGE MEDIAN	DIGESTED SLUDGE GEO. MEAN X SPREAD	DIGESTED SLUDGE MEDIAN
NITROGEN	80,000 x 4.32	32,000	37,100 x 2.64	25,800
PHOSPHORUS	9,070 x 2.04	7,650	16,700 x 1.60	17,800
SULFUR	3,100 x 3.33	5,590	6,010 x 2.09	6,550
% VS (a)	74.4 x 1.12	75	51.9 x 1.24	54
BTU/LB(b)	7,910 x 1.10	8,040	5.850 x 1.14	5,760

(a)PERCENT VOLATILE SOLIDS

(b)HEAT OF COMBUSTION EXPRESSED AS BTU/LB

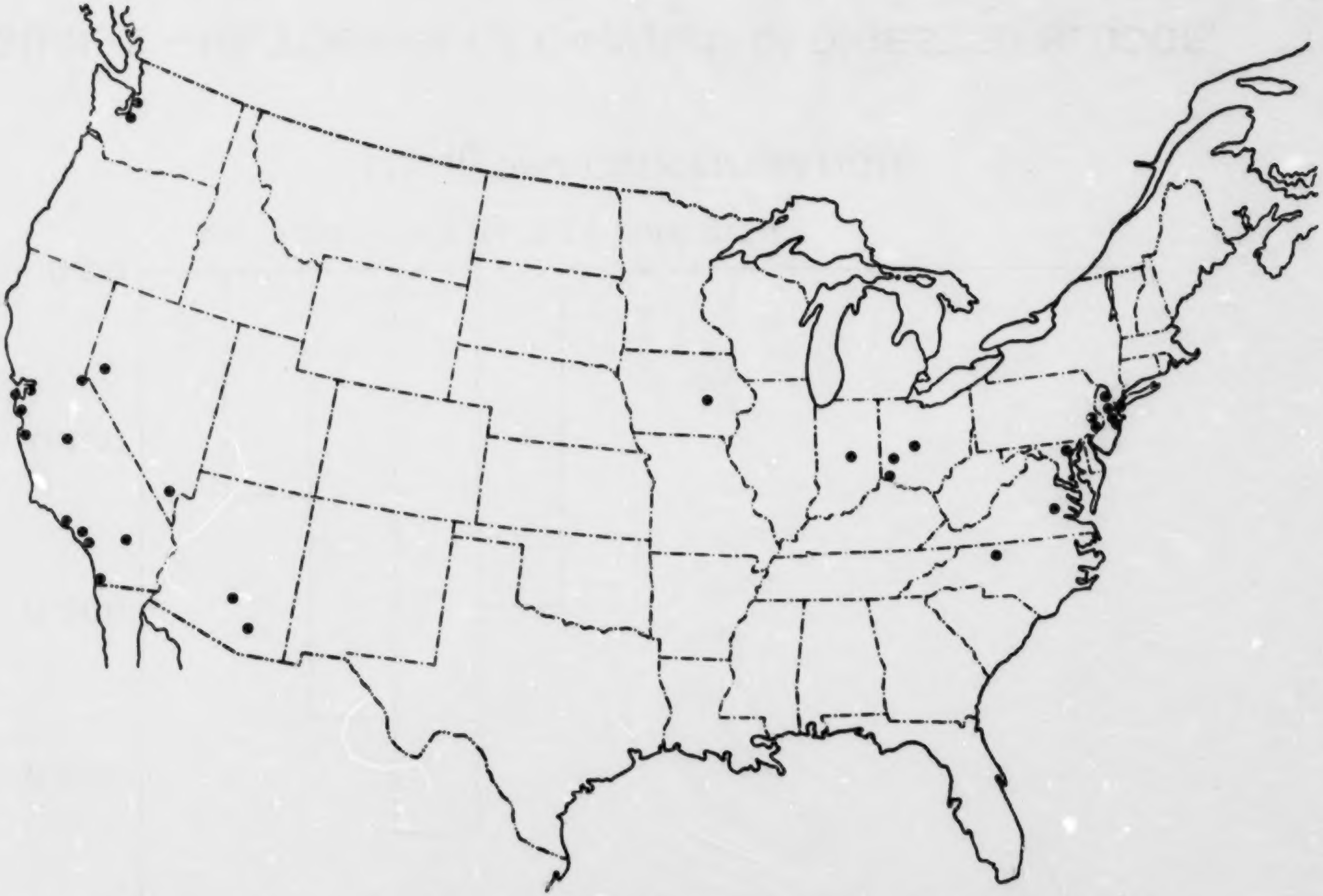


FIGURE 1—LOCATIONS OF WASTEWATER TREATMENT PLANTS IN THE UNITED STATES SUBMITTING SAMPLES.

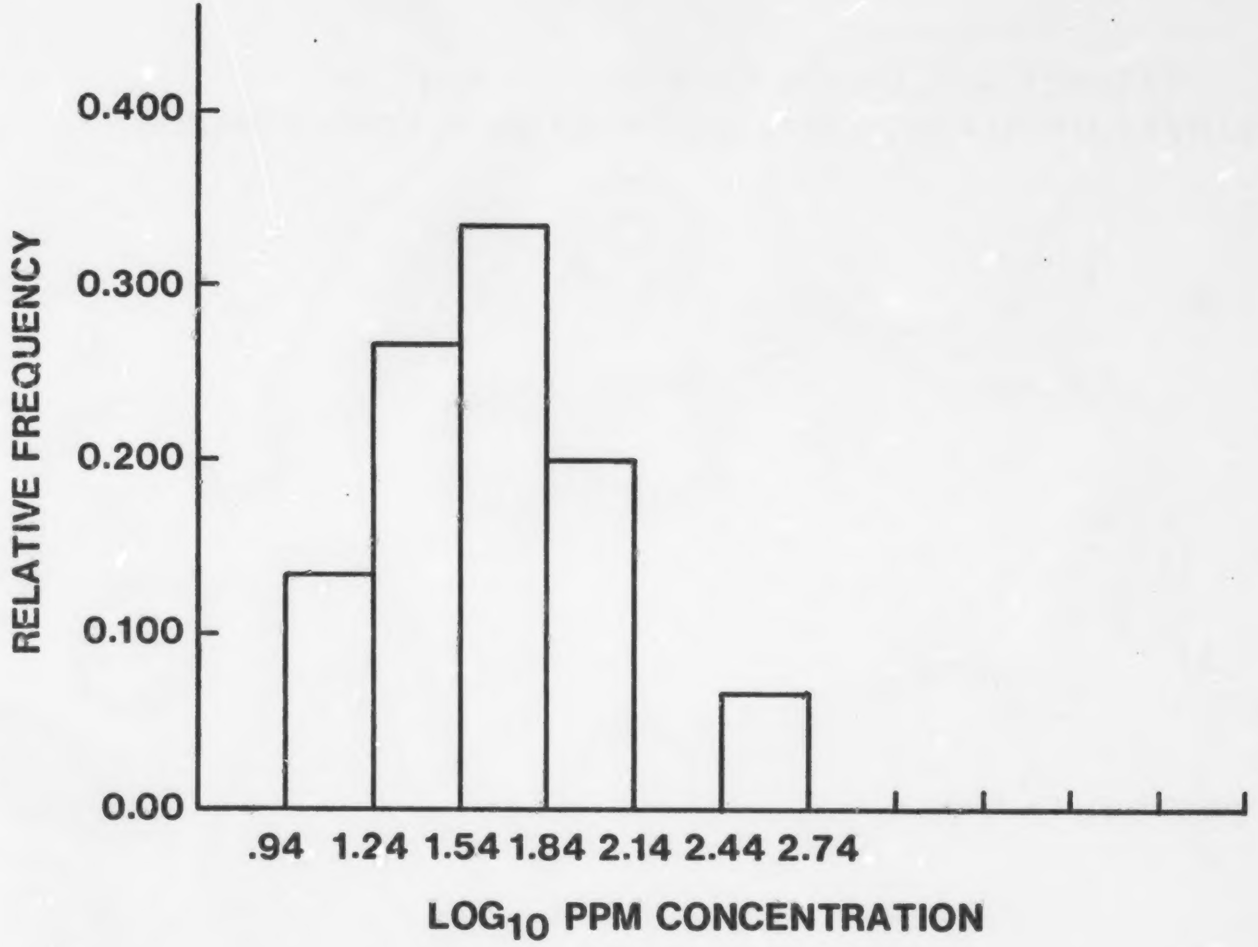


FIGURE 2 —HISTOGRAM OF CADMIUM IN DIGESTED SLUDGE.

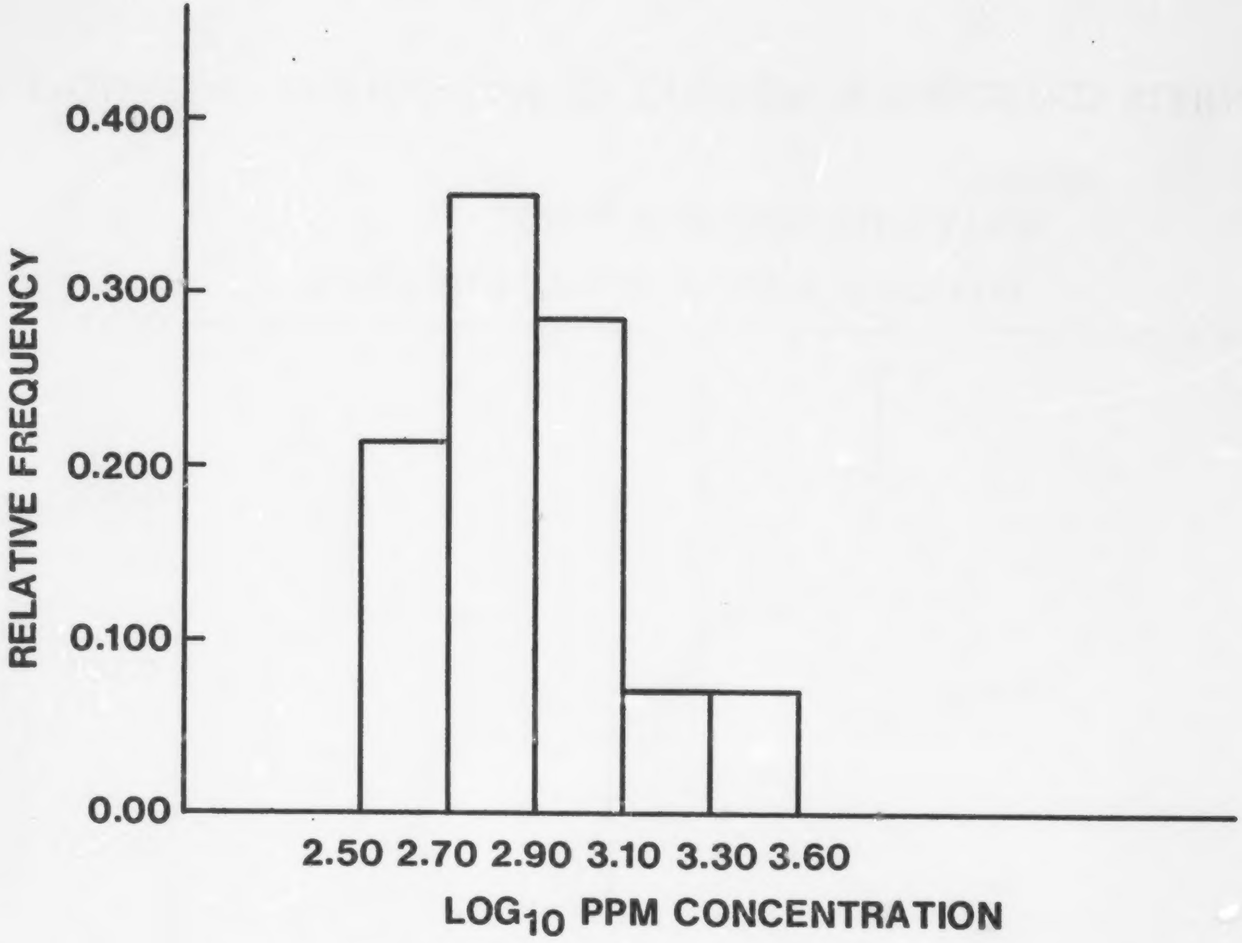


FIGURE 3 —HISTOGRAM OF COPPER IN PRIMARY SLUDGE.

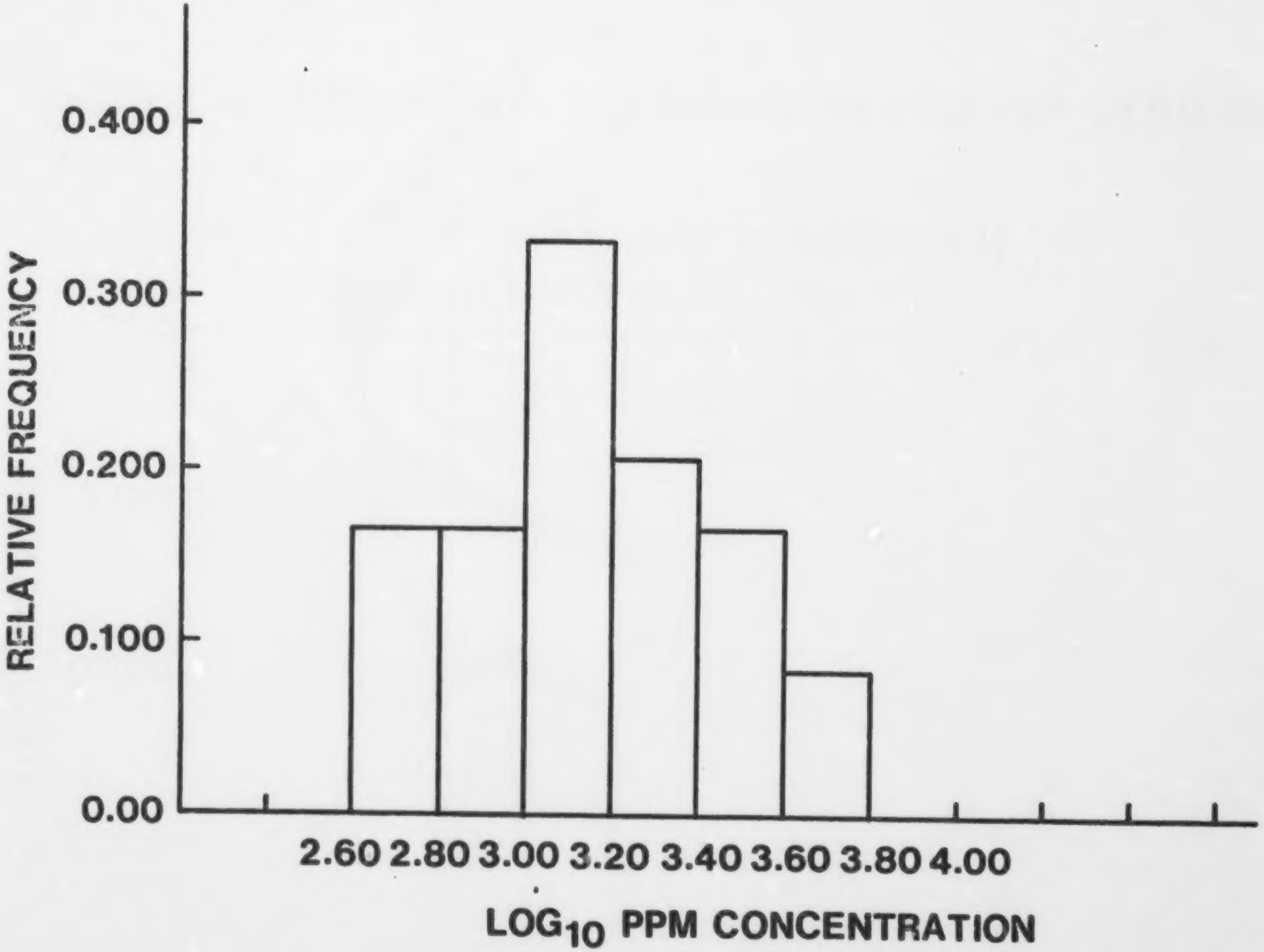


FIGURE 4 —HISTOGRAM OF COPPER IN DIGESTED SLUDGE.

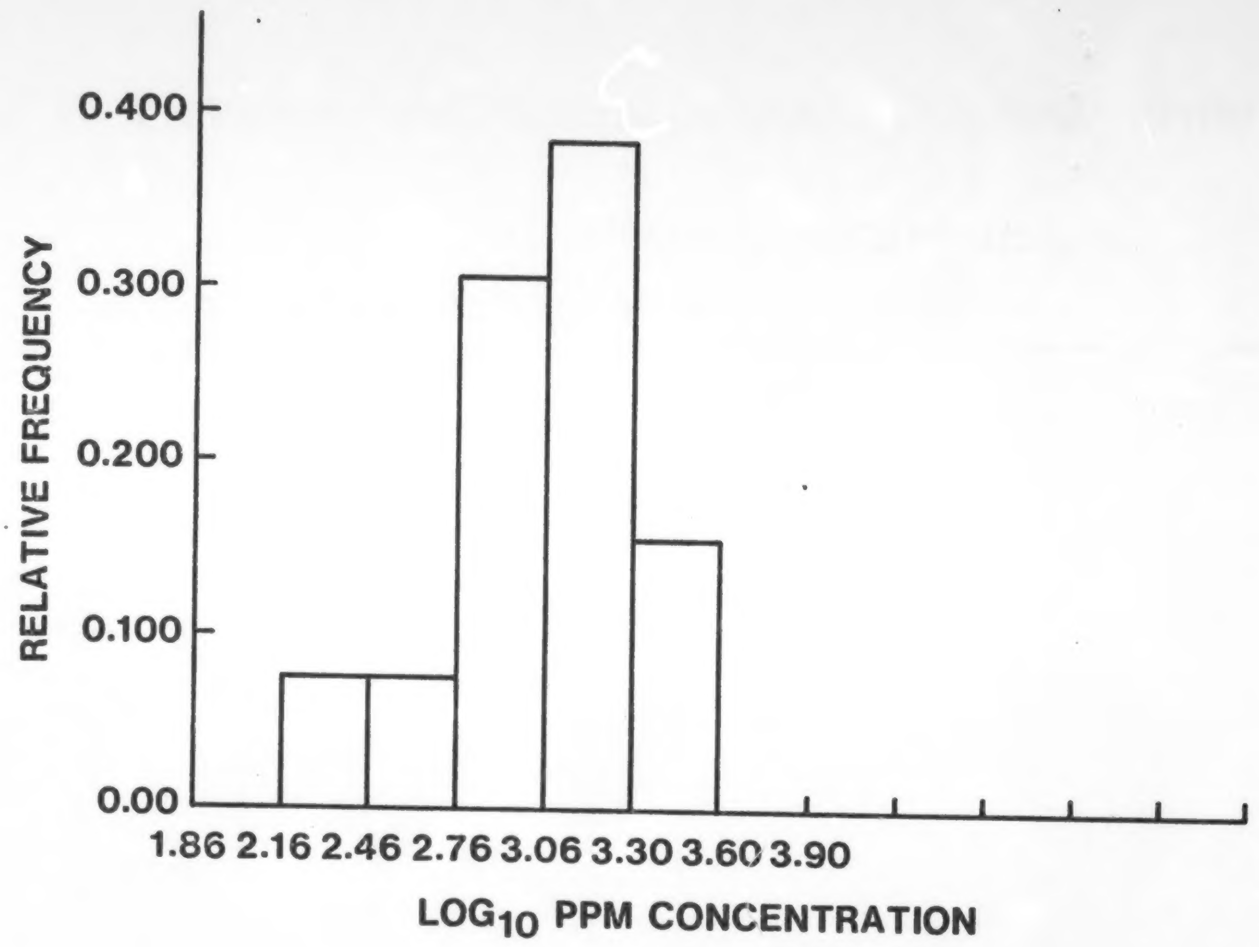


FIGURE 5 –HISTOGRAM OF LEAD IN PRIMARY SLUDGE.

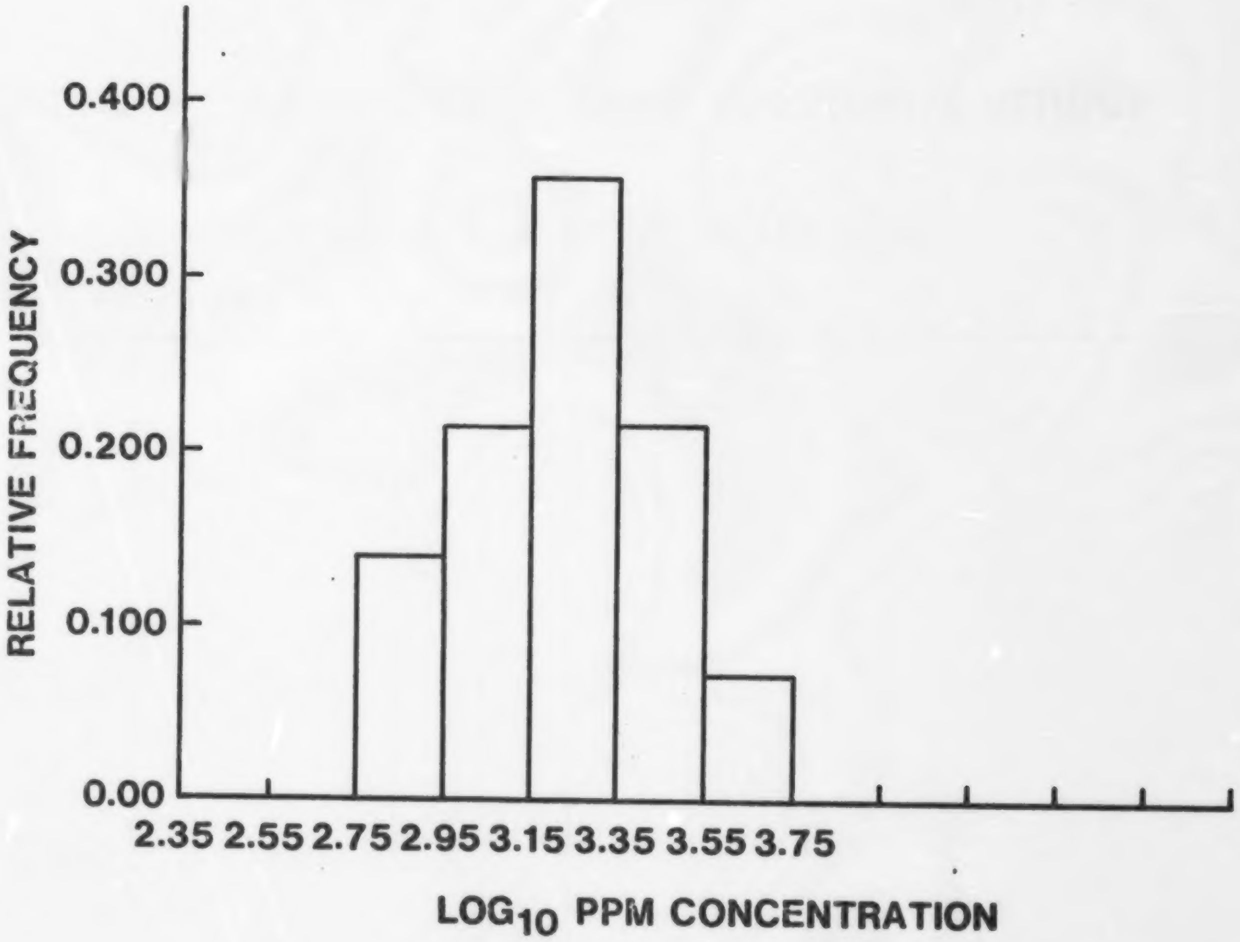


FIGURE 6 —HISTOGRAM OF ZINC IN PRIMARY SLUDGE.

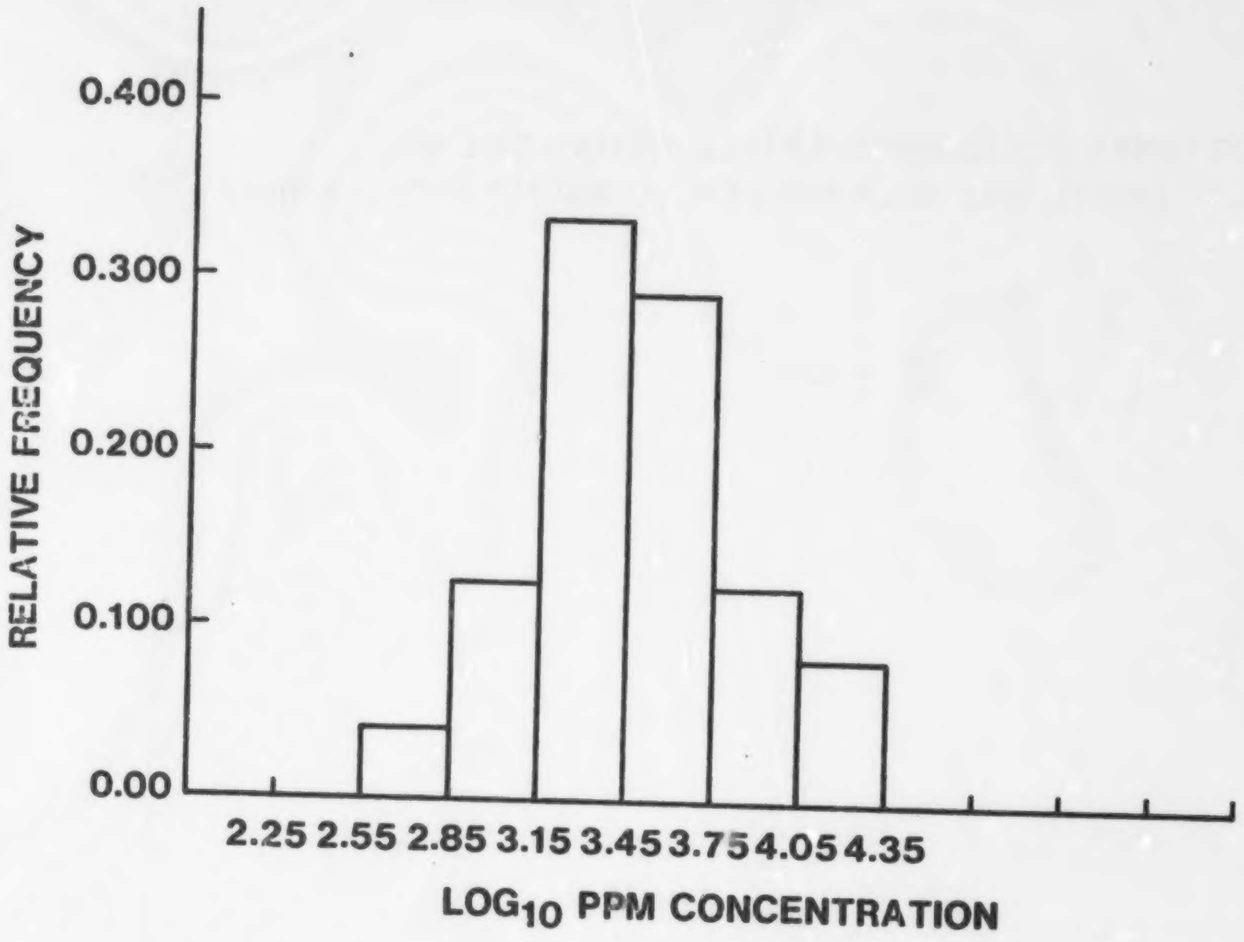


FIGURE 7—HISTOGRAM OF ZINC IN DIGESTED SLUDGE.

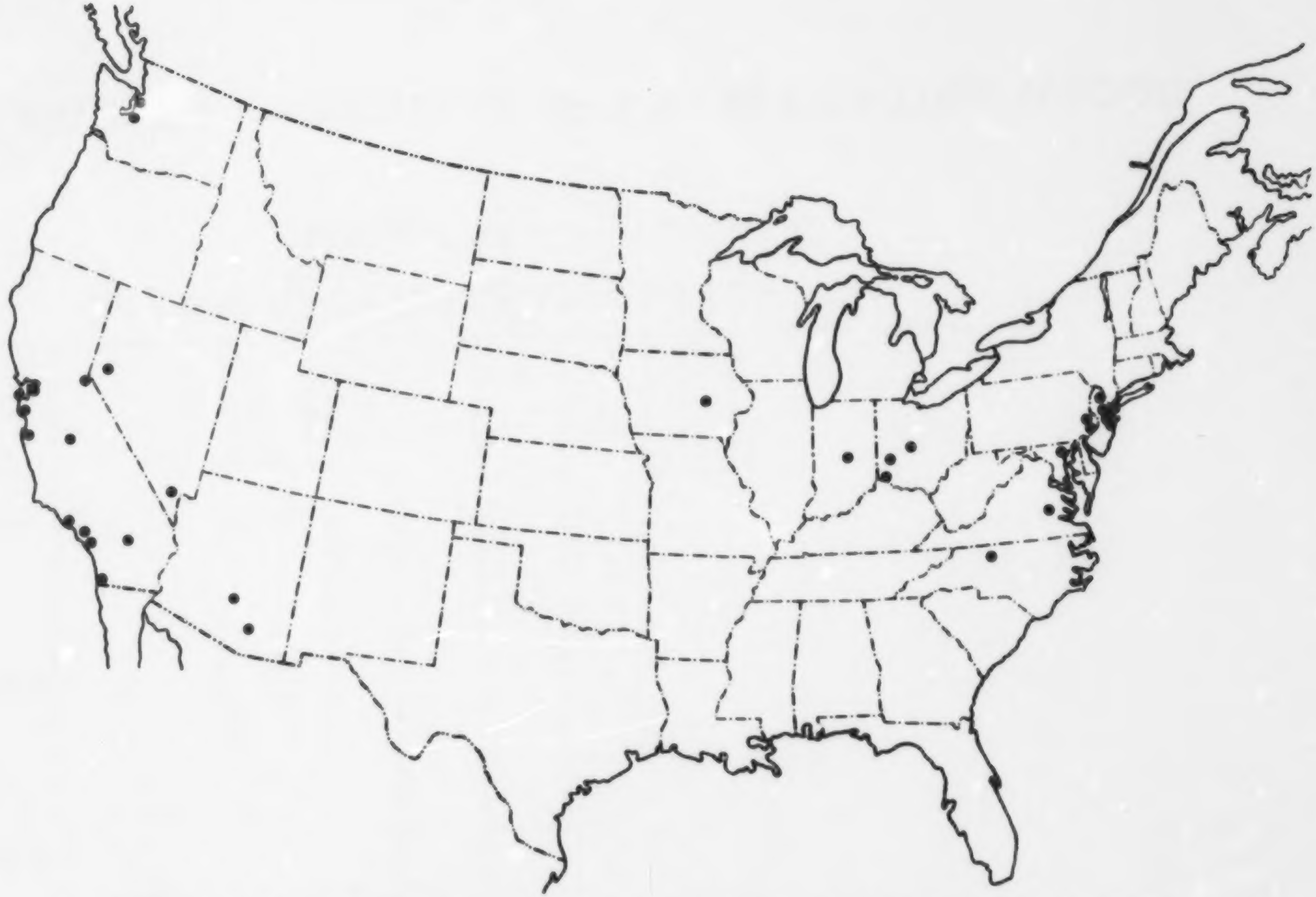


FIGURE 1—LOCATIONS OF WASTEWATER TREATMENT PLANTS IN THE UNITED STATES SUBMITTING SAMPLES.

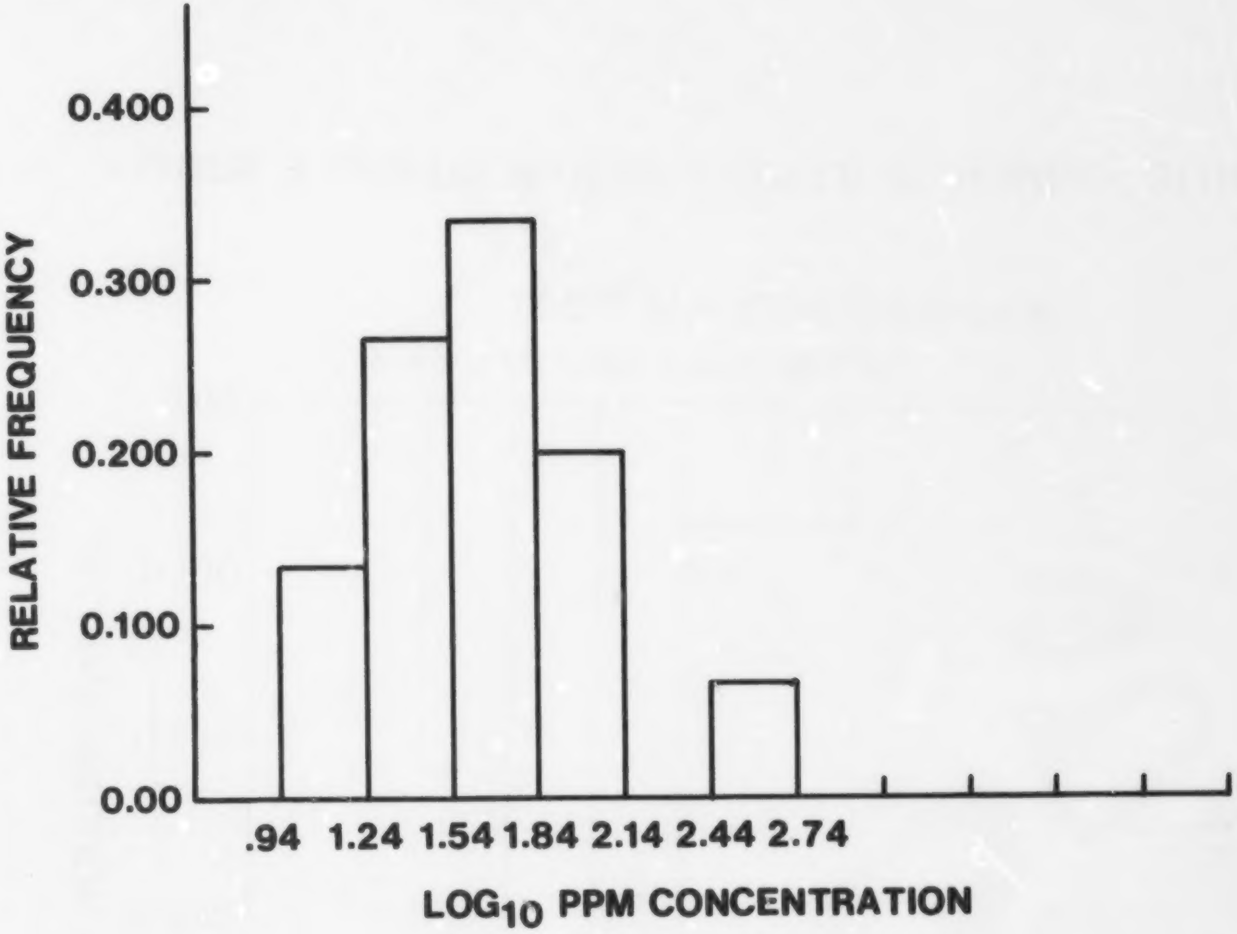


FIGURE 2 —HISTOGRAM OF CADMIUM IN DIGESTED SLUDGE.

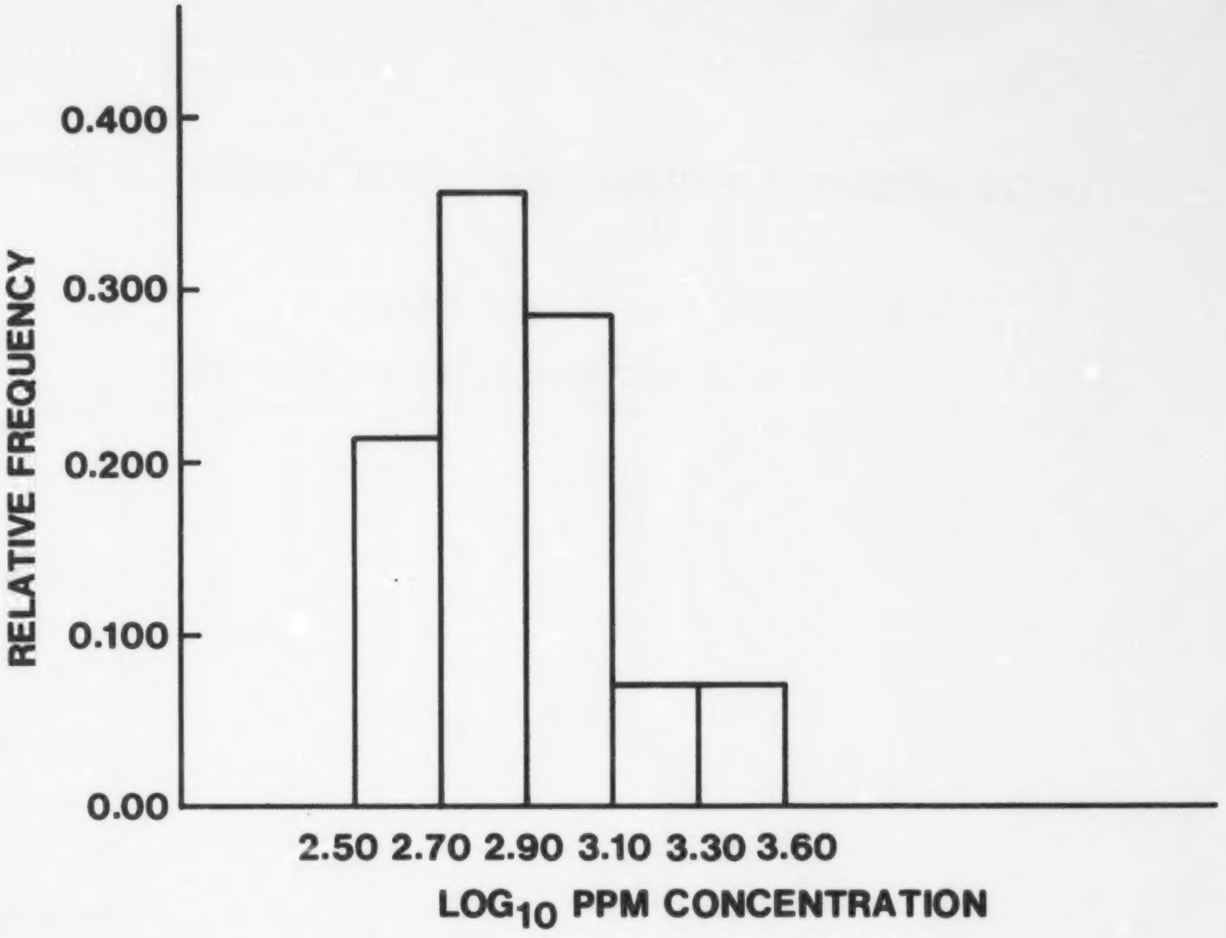
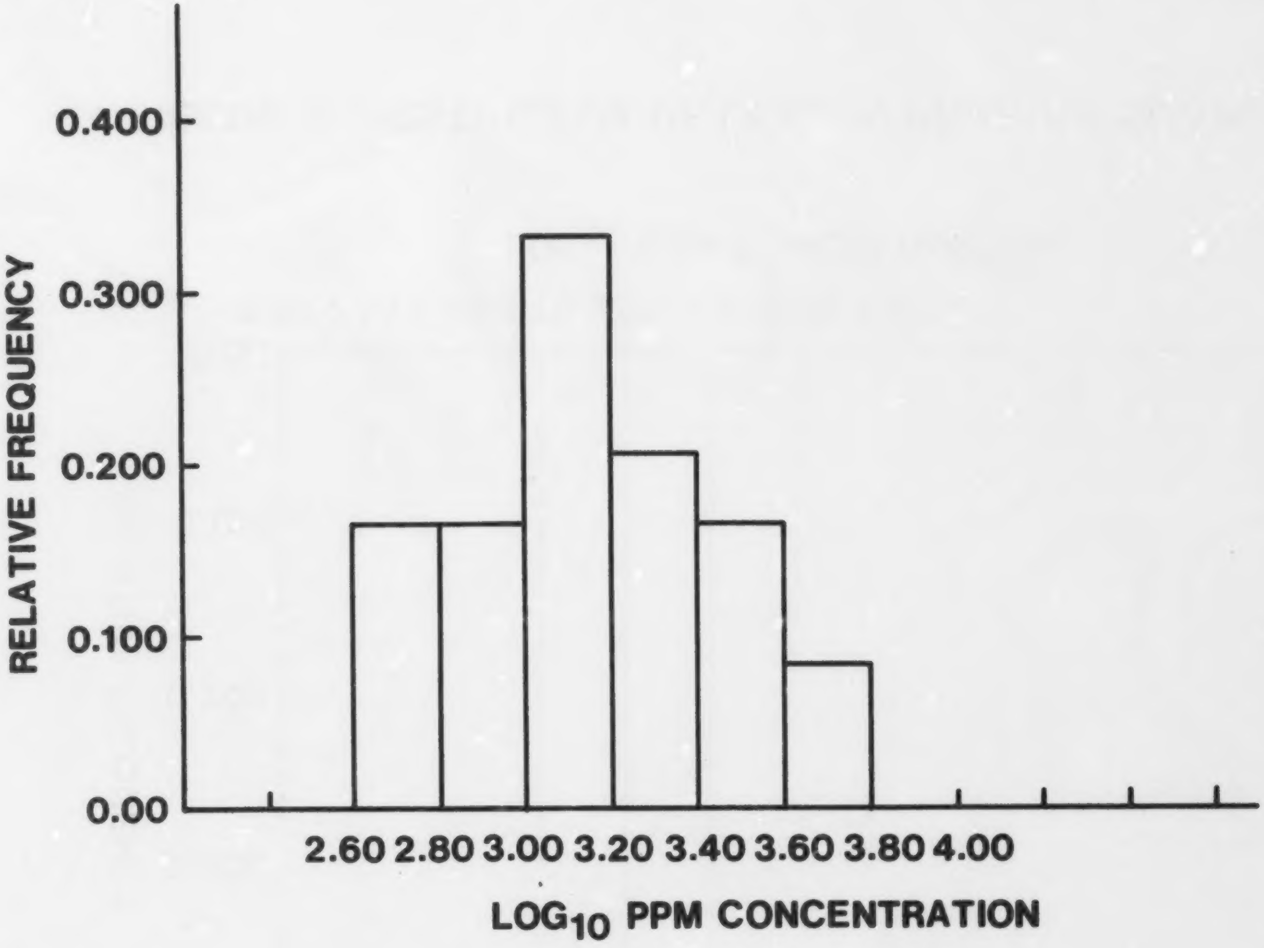


FIGURE 3 —HISTOGRAM OF COPPER IN PRIMARY SLUDGE.



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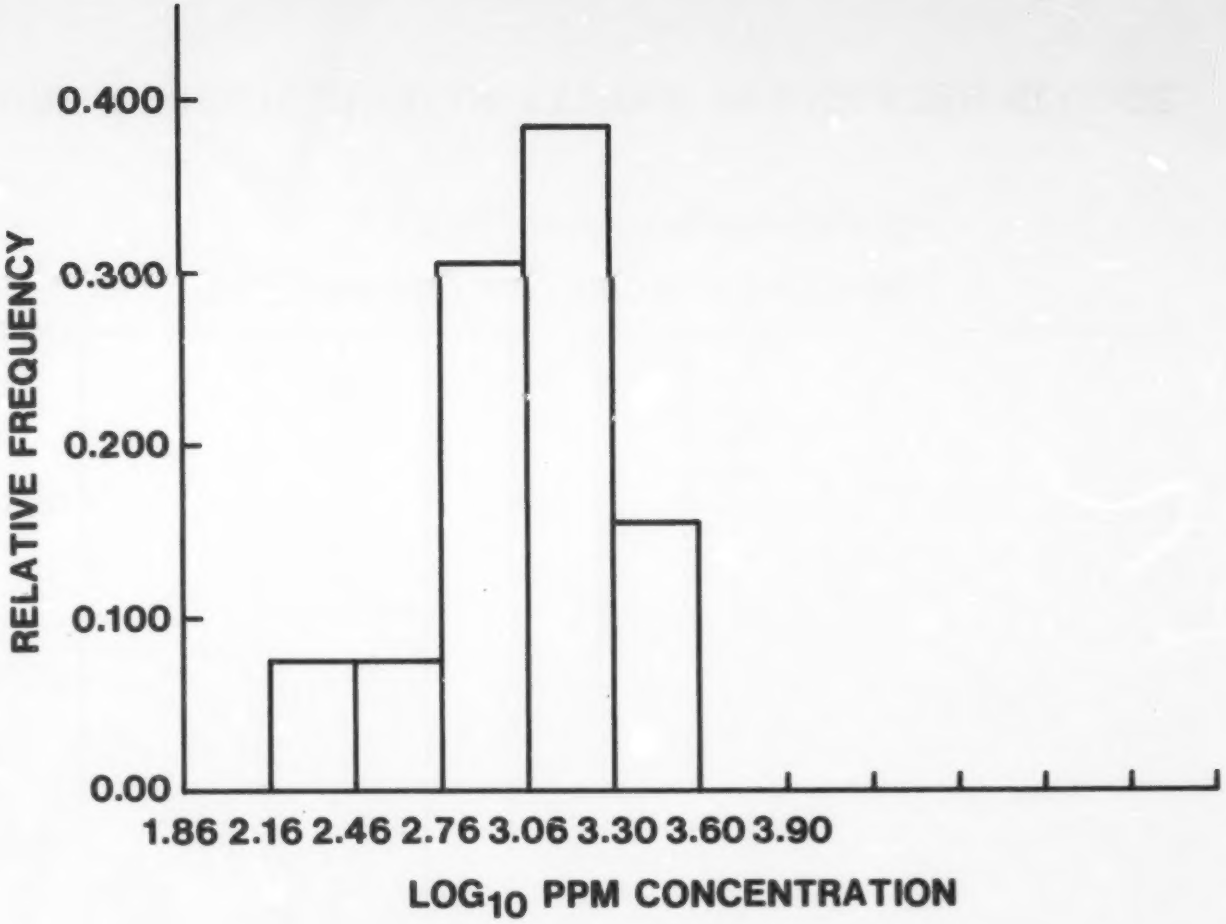

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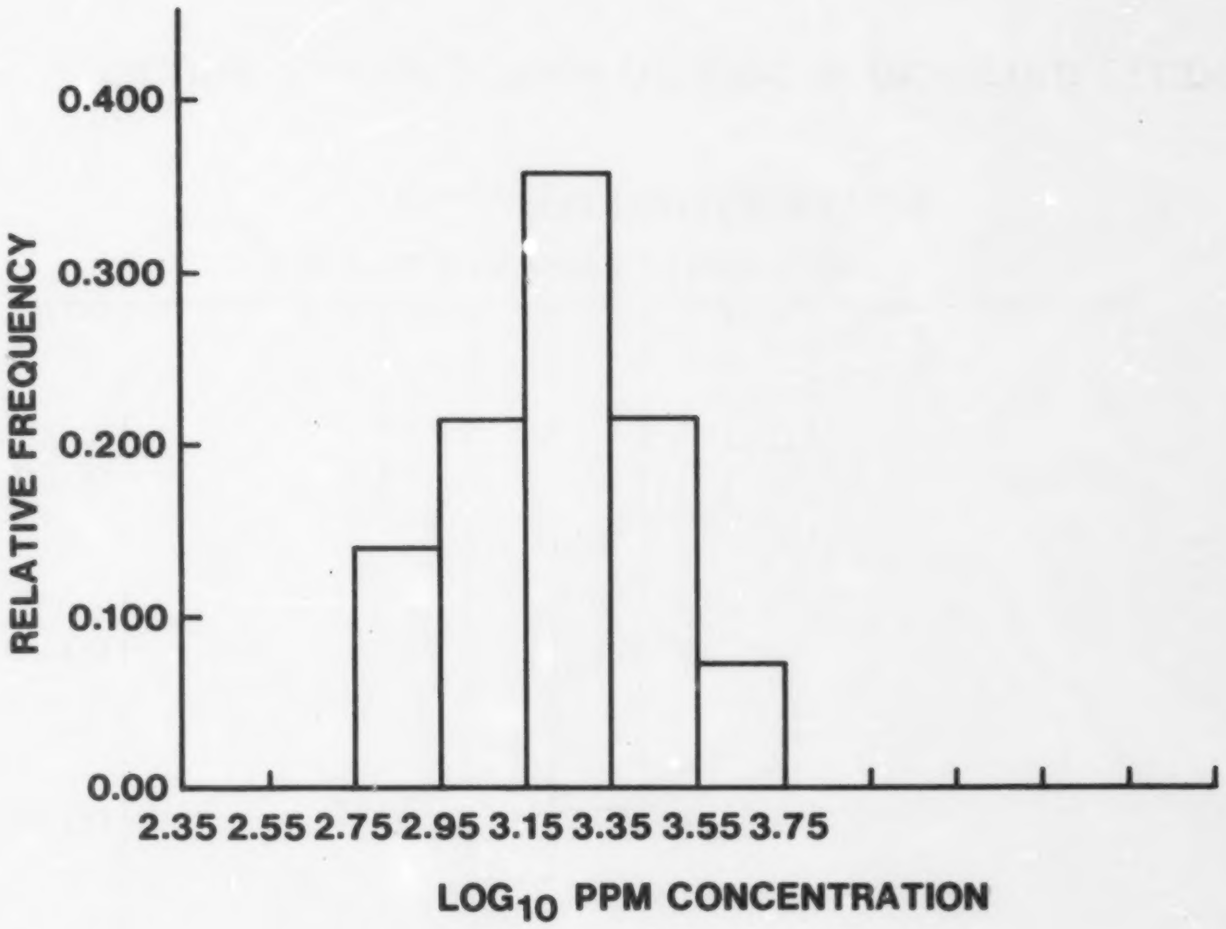

FIGURE 6 —HISTOGRAM OF ZINC IN PRIMARY SLUDGE.

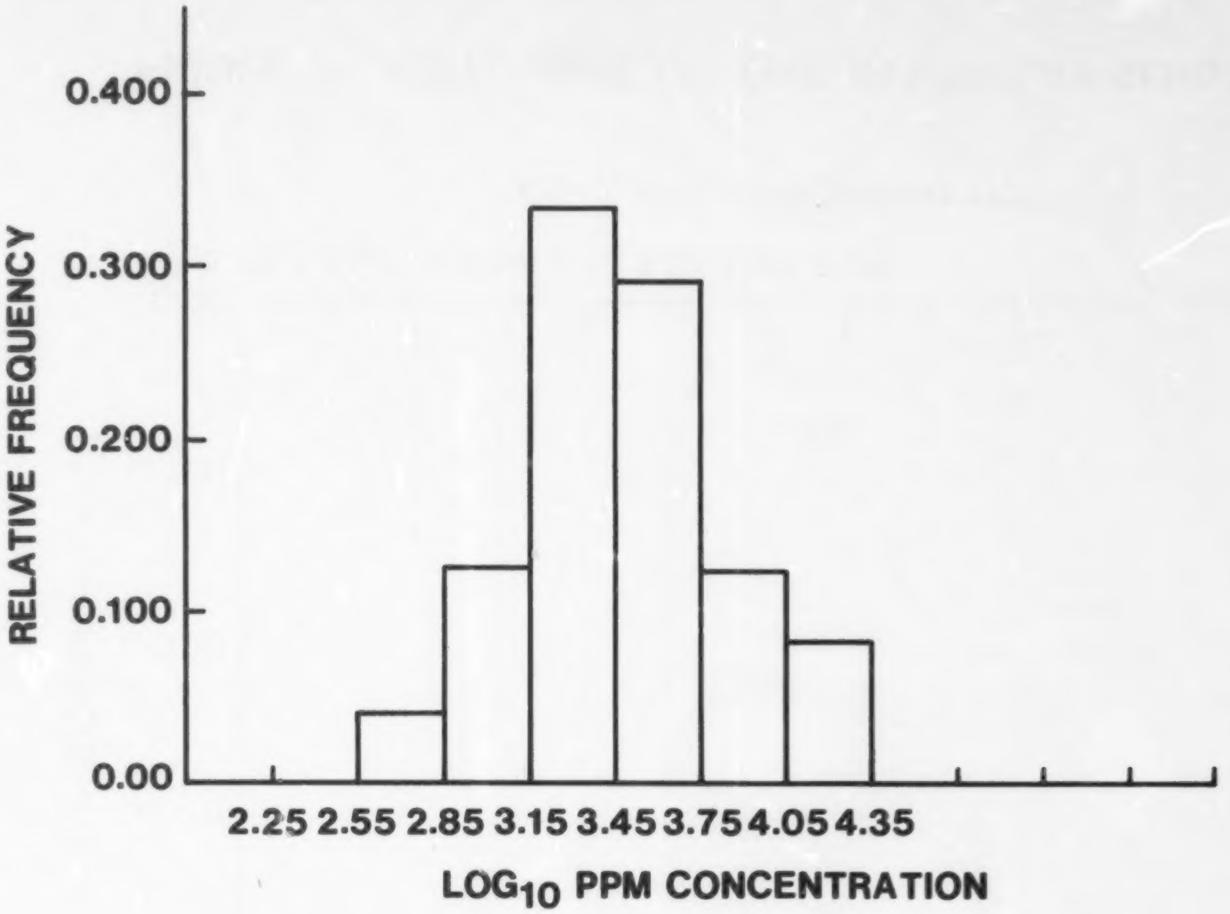

FIGURE 7 — HISTOGRAM OF ZINC IN DIGESTED SLUDGE.

STABILIZATION OF MUNICIPAL SEWAGE SLUDGE BY HIGH LIME DOSE

by

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STABILIZATION OF MUNICIPAL SEWAGE SLUDGE BY HIGH LIME DOSE

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INTRODUCTION

Sludge treatment and ultimate disposal represent a major portion of municipal wastewater treatment costs. Generally efforts in this area have been directed towards minimizing the volume of sludge requiring disposal and reducing the potential for producing nuisance conditions and public health hazards during and after disposal. Processes such as anaerobic digestion, aerobic digestion, and incineration have been used extensively for sludge stabilization. However, each of these processes adds significantly to the cost of wastewater treatment and none totally eliminates the residue which requires disposal.

The use of sewage sludge on cropland is limited by several factors which are of particular concern to environmentalists and public health officials: the nitrogen content of the sludge, the concentration of metals and other trace elements, and the survival of pathogens. Treatment of sludge to reduce its pathogen content and, therefore, its potential for introducing pathogens into cropland was a major concern of this program.

Historically, lime has been used to treat nuisance conditions resulting from open pit privies and the graves of deceased domestic animals. The scope of the program followed from the work of Farrell, et al.,[1] at the Lebanon, Ohio, wastewater treatment plant. In that work, Farrell and his co-workers were concerned with developing a treatment technique for processing that portion of the plant's sludge production which exceeded its digester design flow. In the Lebanon study, lime addition to the sludge was found to be effective in both deodorization and disinfection. The current program was designed as an investigation of the pertinent operating parameters for lime stabilization of sludges and the subsequent effects of application of the lime treated sludges directly to cropped lands.

SCOPE OF THE PROGRAM

The two major objectives of this program were: 1) to determine the degree of stability produced in sludges by the addition of large amounts of lime, and 2) to determine the effects of spreading lime stabilized sludges on land used for crop production. Initial work to achieve the first objective was accomplished through bench scale laboratory studies designed to aid in selection of pilot plant equipment and operational parameters. The majority of the work in this part of the study, however, was conducted on a larger pilot scale. Work on the second objective was accomplished in small scale greenhouse studies and on larger outdoor plots which received varying amounts of sludge. After sludge application, the outdoor plots were cultivated and cropped using standard agricultural techniques.

DISCUSSION OF MAJOR FINDINGS

Laboratory Work

Bench scale laboratory studies were conducted to develop basic information on the lime stabilization process itself and to develop data for use in design and operation of a pilot plant. These bench scale studies were concerned with:

1. lime requirements to achieve specified pH levels within a range from pH 11.0 to 12.4,
2. time dependency of the lime/sludge reaction, and
3. pathogenic bacteria and obnoxious odor reduction as a function of pH and contact time between the lime and the raw sludge.

Unless otherwise noted, all sludge used in the laboratory studies was a mixture of primary sludge and trickling filter humus and was taken from the digester feed line at the Richland, Washington municipal sewage treatment plant.

Laboratory studies were conducted at the beginning of the program to determine the lime dose required to raise the pH to a specified level. The results from these studies were used in design and operation of the pilot plant facility which was used to produce lime stabilized sludges for use on the plots in outdoor growth studies.

The pH levels chosen for investigation were 11.0, 11.2, 11.4, 11.6, 11.8, 12.0, and 12.4. One liter raw domestic sewage sludge samples with known total solids concentrations were dosed with a 100 mg $Ca(OH)_2$/ml lime slurry and mixed with a paddle stirrer until the change in pH reached equilibrium. Lime dose and the resulting pH were then recorded. This procedure was repeated until the specified pH level was reached. Sludges with different total solids concentrations were treated in this same manner to develop data on the lime dose required to raise pH to specified levels in sludges with different solids contents.

The results from these studies are shown in Figure 1. These results indicate that total solids concentration affects the lime dose required to raise the pH to a specified level. As can be seen from Figure 1, the lime requirements increased as total solids concentration increased. This variation in lime requirements is probably caused by a combination of

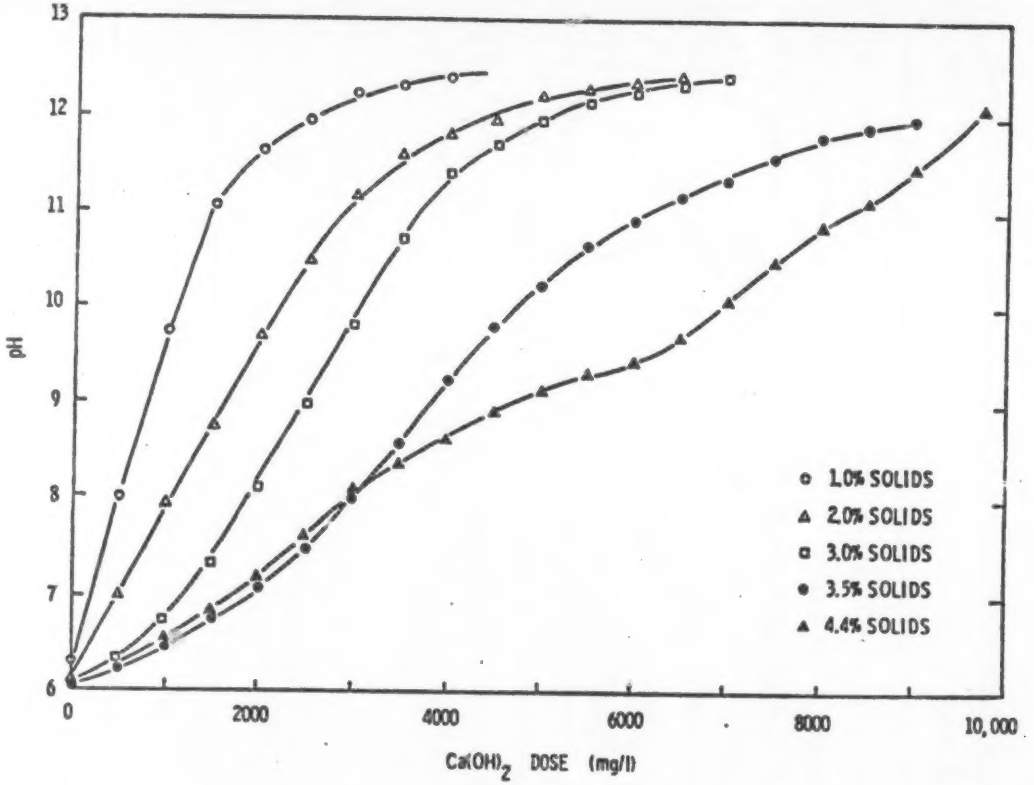

FIGURE 1. LIME DOSES REQUIRED TO RAISE pH IN SLUDGES WITH DIFFERENT SOLIDS CONCENTRATIONS.

factors including: 1) difficulty in establishing good mixing patterns in the thicker sludges, and 2) chemical demand caused by reaction of the hydroxyl ions with dissolved CO_2, bicarbonate alkalinity, and organic materials (neutralizing organic acids, hydrolysis, saponification). A low shear, paddle mixing technique was used to prevent homogenization of the sludge. Difficulty in establishing good mixing patterns in the sample container was encountered with the higher solids concentration sludges. This difficulty with establishing good mixing patterns in the thicker sludges could possibly have prevented intimate contact between the lime slurry and the liquid phase component of the sludge. Thus, a dissolution of $Ca(OH)_2$ introduced into the sludge would be hindered and more lime would be required to elevate the pH of the system. The lime demand would also increase as solids content increased since more organic matter would be introduced with a concomitant increase in the hydroxyl ion requirement for neutralizing organic acids and reactions involving hydrolysis and saponification.

From this discussion, it appears that the lime dose required to raise the pH of a given sludge to a specified level would be significantly influenced by the chemical characteristics and the solids concentration of the sludge and by the technique used to mix the lime and sludge.

Previous work on lime-sludge systems has shown that a pH decay is experienced as the treated sludge ages.[1,2,3] Decay from high pH levels to lower levels can change the system environment from one hostile to microbial survival to one suitable for organism existence and growth. Therefore, laboratory studies were undertaken to define the extent of pH decay experienced in sludges after lime treatment.

A sludge sample with total solids concentrations of 4.4 percent was collected and divided into one liter batches which were then lime treated to pH levels of 11.0, 11.2, 11.4, 11.6, 11.8, 12.0, and 12.4. pH decay in each of these samples was monitored over a 24 hour time period. Results from this study are shown in Figure 2.

As can be seen from the results, pH decay was observed in all samples as the lime treated sludges aged. However, the degree of decay significantly decreased when the initial value of a sample was 12.0 or greater.

This decay is believed to be caused by the sludge chemical demand exerted on the hydroxyl ions supplied in the lime slurry. Many of the reactions which exert this demand probably proceed slowly in this type of system (non-optimal chemical reactor), and thus pH decays slowly as hydroxyl ions enter into chemical reactions. The degree of decay probably decreases as initial pH increases because of the extremely large quantities of lime required to elevate pH to 12.0 or greater. Large concentrations of both hydroxyl ions and undissociated $Ca(OH)_2$ are supplied in the slurry. Therefore, at high pH, sufficient OH^- species are present in the system to allow chemical reactions to proceed without an attendant decrease in pH. In summary, pH decay depends upon both the quantity of lime added and the total solids concentration of the sludge.

A major portion of this program was concerned with definition of the effects of high lime dose on the pathogen populations in sewage sludges. Therefore, a laboratory study was conducted in which mixed primary and secondary sludges with a total solids content of 4.4 percent was lime treated to various pH levels within the 11.0 to 12.4 range. Laboratory beakers containing the lime treated sludge were allowed to stand open to the atmosphere at room temperature, and

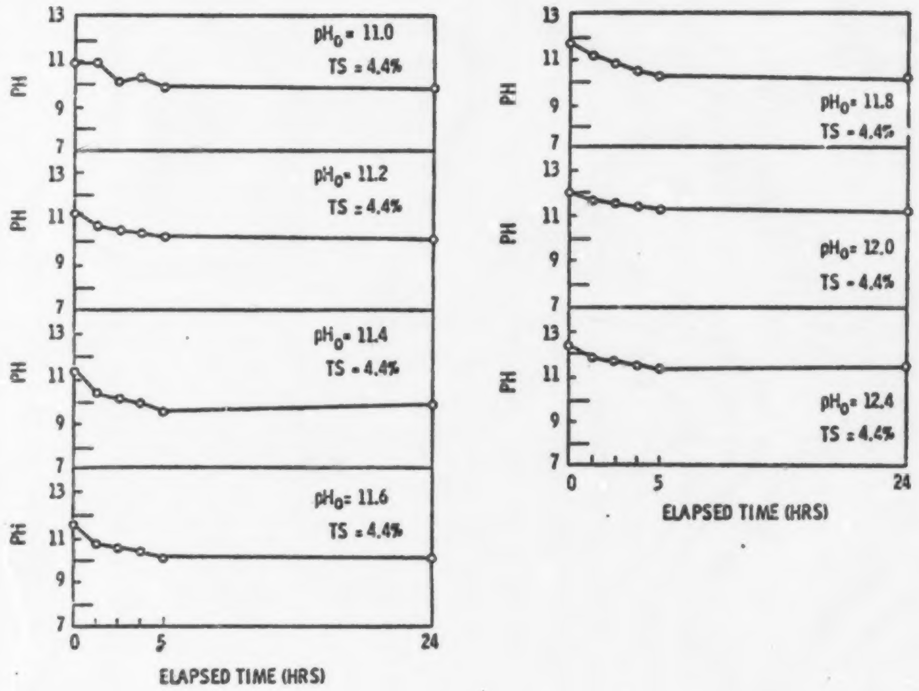

FIGURE 2. LIME-SLUDGE pH REACTION TIME DEPENDENCY FOR 4.4% SLUDGE SOLIDS

samples for bacteriological analysis were collected after lime-sludge contact times of one and twenty-four hours. The microorganisms chosen as indicators of pathogen response to lime treatment were fecal coliform, fecal streptococci, *Salmonella* species, and *Pseudomonas aeruginosa*. Bacteriological methods used for determination of *Salmonella* species and *Pseudomonas aeruginosa* were developed by Kenner, et al.[4] The membrane filter technique and plate count technique, both described in Standard Methods,[5] were used to count fecal coliforms and fecal streptococci, respectively.

Results from these studies are shown in Tables 1 and 2. After one hour of contact time, pathogen reductions were observed in the lime treated sludges at all pH values within the range under study. In general, the degree of reduction increased as pH increased with consistently high pathogen reductions occurring only after the pH reached 12.0. Fecal streptococci appeared to resist inactivation by lime treatment particularly well at the lower pH values in the study range. However, at pH 12.0 these organisms were also inactivated after one hour of contact time.

An increase in pathogen counts was usually observed in the samples taken after twenty-four hours of contact time. It should also be noted that the pH of the lime-sludge system usually decreased during this time period. Work done by Paulsrud and Eikum[3] on lime stabilization of sewage sludges in Norway showed that pH could be maintained at high levels by overdosing the system with $Ca(OH)_2$. This procedure provides surplus lime to the system so that chemical demand does not cause a significant decrease in pH. For sludge which is to be spread on agricultural land, addition of excess quantities of lime in some cases might harm crop production.

TABLE 1

EFFECT OF LIME ON FECAL COLIFORM AND FECAL STREPTOCOCCI AT 4.4% SLUDGE SOLIDS CONCENTRATION

Initial pH Value	Lime Contact Time (hrs)	pH When Sample Taken	Fecal Coliform per 100 ml	Fraction of Original Remaining	Fecal Streptococci per 100 ml	Fraction of Original Remaining
6.8*			1.60×10^{7}		4.13×10^{7}	
6.8*			2.03×10^{7}		4.67×10^{7}	
11.0	1	10.3	3.65×10^{4}	2.01×10^{-3}	5.37×10^{7}	1.22
11.0	1	10.3	4.35×10^{4}	2.40×10^{-3}	6.50×10^{7}	1.47
11.0	24	8.6	2.87×10^{4}	1.58×10^{-3}	7.33×10^{7}	1.66
11.0	24	8.6	3.25×10^{4}	1.79×10^{-3}	7.50×10^{7}	1.70
11.2	1	10.6	3.72×10^{4}	2.05×10^{-3}	8.50×10^{7}	1.93
11.2	1	10.6	2.70×10^{4}	1.49×10^{-3}	8.50×10^{7}	1.93
11.2	24	8.7	1.70×10^{4}	9.37×10^{-4}	7.70×10^{7}	1.75
11.2	24	8.7	1.37×10^{4}	7.55×10^{-4}	7.50×10^{7}	1.70
11.4	1	10.7	1.63×10^{4}	8.90×10^{-4}	6.43×10^{7}	1.46
11.4	1	10.7	2.10×10^{4}	1.16×10^{-3}	6.57×10^{7}	1.49
11.4	24	9.2	3.20×10^{4}	1.76×10^{-3}	8.47×10^{7}	1.93
11.4	24	9.2	4.95×10^{4}	2.09×10^{-3}	8.57×10^{7}	1.95
11.6	1	11.1	5.15×10^{4}	2.84×10^{-3}	4.47×10^{6}	0.10
11.6	1	11.1	5.03×10^{4}	2.77×10^{-3}	4.53×10^{6}	0.10
11.6	24	9.5	2.55×10^{4}	1.40×10^{-3}	8.30×10^{7}	1.89
11.6	24	9.5	3.80×10^{4}	2.09×10^{-3}	8.33×10^{7}	1.89
11.8	1	11.5	6.65×10^{4}	3.66×10^{-3}	8.33×10^{6}	0.19
11.8	1	11.5	3.48×10^{4}	1.92×10^{-3}	8.50×10^{6}	0.19
11.8	24	9.9	5.34×10^{4}	2.94×10^{-3}	8.27×10^{6}	0.19
11.8	24	9.9	5.77×10^{4}	3.18×10^{-3}	8.70×10^{6}	0.20
12.0	1	11.8	3.70×10^{4}	2.04×10^{-3}	3.27×10^{5}	7.43×10^{-3}
12.0	1	11.8	2.25×10^{4}	1.24×10^{-3}	3.37×10^{5}	7.66×10^{-3}
12.0	24	10.6	1.15×10^{4}	6.34×10^{-3}	3.09×10^{6}	0.07
12.0	24	10.6	2.20×10^{4}	1.21×10^{-3}	2.50×10^{6}	0.06
12.4	1	12.1	2.70×10^{4}	1.49×10^{-3}	7.20×10^{5}	0.02
12.4	1	12.1	5.35×10^{4}	2.95×10^{-3}	7.87×10^{5}	0.02
12.4	24	11.4	7.13×10^{4}	3.93×10^{-3}	2.61×10^{6}	0.06
12.4	24	11.4	7.78×10^{4}	4.28×10^{-3}	2.55x106	0.06

*Raw Sludge

TABLE 2

EFFECT OF LIME ON SALMONELLA SPECIES AND PSEUDOMONAS AERUGINOSA AT 4.4% SLUDGE SOLIDS CONCENTRATION

Initial pH Value	Lime Contact Time (hrs)	pH When Sample Taken	Samonella Species per 100 ml	Fraction of Original Remaining	Pseudomonas Aeruginosa per 100 ml	Fraction of Original Remaining
6.8*			10,800		2800	
6.8*			5600		8600	
11.0	1	10.3	1080	0.13	144	0.03
11.0	1	10.3	220	0.03	128	0.02
11.0	24	8.6	340	0.04	7000	1.23
11.0	24	8.6	340	0.04	1400	0.25
11.2	1	10.6	340	0.04	28	4.91×10^{-3}
11.2	1	10.6	340	0.04	28	4.91×10^{-3}
11.2	24	8.7	440	0.05	9	1.58×10^{-3}
11.2	24	8.7	260	0.03	9	1.58×10^{-3}
11.4	1	10.7	22	2.68×10^{-3}	100	0.02
11.4	1	10.7	320	0.04	94	0.02
11.4	24	9.2	560	0.07	1840	0.32
11.4	24	9.2	260	0.03	62	0.01
11.6	1	11.1	44	5.37×10^{-3}	14	2.39×10^{-3}
11.6	1	11.1	98	0.01	16	2.74×10^{-3}
11.6	24	9.5	220	0.03	8	1.40×10^{-3}
11.6	24	9.5	52	6.34×10^{-3}	4	7.02×10^{-4}
11.8	1	11.5	260	0.03	4	7.02×10^{-4}
11.8	1	11.5	98	0.01	4	7.02×10^{-4}
11.8	24	9.9	220	0.03	9	1.58×10^{-3}
11.8	24	9.9	34	4.14×10^{-3}	9	1.58×10^{-3}
12.0	1	11.8	4	4.88×10^{-4}	0	0.00
12.0	1	11.8	9	1.10×10^{-3}	0	0.00
12.0	24	10.6	28	3.41×10^{-3}	0	0.00
12.0	24	10.6	28	3.41×10^{-3}	8	1.40×10^{-3}
12.4	1	12.1	16	1.90×10^{-3}	0	0.00
12.4	1	12.1	14	1.66×10^{-3}	0	0.00
12.4	24	11.4	40	4.88×10^{-3}	26	4.56×10^{-3}
12.4	24	11.4	66	8.05×10^{-3}	26	3.86×10^{-3}

---

*Raw Sludge

An important factor in any stabilization process is the ability to significantly reduce the obnoxious odor producing potential of the sludge. Odors usually result from anaerobic decomposition of the sludge organic content. Conventional methods of reducing the odor producing potential in sludge are based on controlled biochemical degredation of the sludge organic matter (aerobic and anaerobic digestion) or total destruction of the organic matter (incineration). The lime stabilization process achieves reductions in odor producing potential by creating a high pH, hostile environment in the sludge, thus eliminating or suppressing the growth of microorganisms that could produce nuisance conditions.

Tests to quantitatively measure odor are subject to inaccuracies since test panels of supposedly unbiased, randomly selected people are usually required. However, since no other standard tests were available, the threshold odor number test described in Standard Methods[5] was used in this study to measure odor in raw and lime treated sludges. Threshold odor number is defined as the greatest dilution of the sample with odor-free water which yields the least perceptible odor. The tests were conducted on mixtures of primary and secondary sludge which had been lime treated to pH levels of 11.0, 11.2, 11.4, 11.6, 11.8, 12.0, and 12.4. The threshold odor numbers of the treated samples were compared to those of sludge samples which had received no treatment. Sludge samples with total solids concentrations of 2.0 and 4.4 percent were used. Samples were tested after one and twenty-four hour contact times.

The results from this study are shown in Table 3. In both cases, the threshold odor number of the raw sludges was found to be 8000 while that of the treated samples usually ranged from 800 to 1330. This data indicates that lime treatment does have a deodorizing effect. Qualitative

TABLE 3

THRESHOLD ODOR NUMBERS FOR TREATED AND UNTREATED SLUDGES WITH DIFFERENT SOLIDS CONCENTRATIONS

Sludge Type and pH Level	Threshold Odor Numbers			
	1 Hr Contact	% Reduction	24 Hr Contact	% Reduction
Total Solids=2.0%				
6.8 (Untreated)	8000		8000	
11.0	1000	88	8000	0
11.2	1000	88	1000	88
11.4	1000	88	1000	88
11.6	1000	88	1000	88
11.8	1000	88	1000.	88
12.0	1000	88	1000	88
12.4	1000	88	1000	88
Total Solids=4.4%				
6.8 (Untreated)	8000		8000	
11.0	1330	83	4000	50
11.2	1330	83	1330	83
11.4	1330	83	4000	50
11.6	1330	83	1330	83
11.8	800	90	800	90
12.0	800	90	1330	83
12.4	1330	83	1330	83

observations in the laboratory substantiate this finding. The intense putrid odors liberated from the sludge samples at the commencement of each test changed to relatively innocuous humus-like odors after lime treatment.

This deodorizing effect is not permanent, however. Surplus amounts of lime added to the sludge can retard pH decay and reoccurance of nuisance conditions. Further, if the lime stabilized sludge is incorporated into the soil, odors should no longer be a problem.

Pilot Plant Work

After development of pilot plant design and operational parameters, construction of the pilot facility commenced. The process flow scheme was quite simple since it basically consisted of a sludge-lime mixing vessel and contact tank to provide the desired contact time. Process control was maintained by periodically monitoring pH of the discharge from the sludge-lime contactor. Air diffusion mixing was used to eliminate the blade fouling problems which were observed during laboratory studies.

Sludge flows ranging from 3 to 5 gpm were treated during pilot plant operations. Lime dose required to achieve the desired sludge pH was monitored routinely and recorded. This allowed optimization of lime feed to minimize process operating costs from the lime usage standpoint. For the most part, influent consisted of a mixture of primary and secondary sludge from the Richland, Washington municipal trickling filter plant. This mixture of sludges was pumped directly from the line which feeds the Richland plant's anaerobic digesters. Additional work was carried out using raw primary sludge and trickling filter secondary sludge separately, and on mixed sludges prethickened with gravity settling.

The results from studies to determine the lime dose required to achieve a specified pH level, pathogen reductions resulting from lime treatment, and the effect of lime treatment on sludge thickening characteristics will be discussed.

In order to optimize chemical feed during pilot plant operations, the lime dose applied to the raw sludge was varied and system pH response was observed. The system was allowed to come to equilibrium after each dose change and pH was recorded. The results from this study are shown in Table 4.

Average system pH for the series of daily rnns ranged from 12.2 through 12.4, and at no time during the runs did pH fall below the desired 12.0 level. The average lime dose ranged from 4.2 to 5.7 g $Ca(OH)_2$ per liter of sludge, and the average overall pilot plant studies was 4.9 g/ℓ. The daily average lime dose expressed as grams $Ca(OH)_2$ per kilogram of raw sludge total solids ranged from 102.2 through 207.8, and the overall average was 141.9. These lime doses are considered the minimum required to maintain pH at or above the desired level ($pH \geq 12.0$) during sludge processing. However, since excess lime was not added to the system, slight pH decay with time would be expected to occur. Paulsrud and Eikum[3] determined that the lime dose required to maintain sludge pH greater than 11.0 for fourteen days varied considerably with the type sludge being treated and prior chemical treatment.

Regression analysis of pilot plant operating data resulted in the following equation which related the required lime dose and the sludge total solids concentration:

$$\text{Lime Dose} = 4.2 + 1.6\,(TS)$$

where: Lime Dose is expressed as grams $Ca(OH)_2$ per liter of sludge

TS = total solids fraction in the sludge

TABLE 4

SUMMARY OF PILOT PLANT OPERATING DATA

No. of Process Control Checks Made	July 24			July 25			July 26			July 31			August 1		
	$Ca(OH)_2$ Dose			$Ca(OH)_2$ Dose			$Ca(OH)_2$ Dose			$Ca(OH)_2$ Dose			$Ca(OH)_2$ Dose		
	g/ℓ	g/kg*	pH	g/ℓ	g/kg*	pH	g/ℓ	g/kg*	pH	g/ℓ	g/kg*	pH	g/ℓ	g/kg*	pH
1	2.4	61.5	12.0	4.9	144.1	12.4	4.2	120.0	12.0	4.8	123.1	12.2	4.6	135.3	12.3
2	3.8	97.4	12.1	4.9	144.1	12.3	4.3	122.9	12.1	5.1	130.8	12.3	4.8	141.2	12.3
3	4.2	107.7	12.2	4.9	144.1	12.3	4.4	125.7	12.2	5.4	138.5	12.3	4.7	138.2	12.3
4	4.8	123.1	12.3	4.9	144.1	12.3	4.8	137.1	12.2	5.8	148.7	12.3	4.7	138.2	12.3
5	4.6	117.9	12.3	4.8	141.2	12.3	5.2	148.6	12.3	5.6	143.6	12.4	4.7	138.2	12.3
6	4.9	125.6	12.3	4.8	141.2	12.4	5.2	148.6	12.3	5.4	138.5	12.4			
7				4.5	132.4	12.4	5.4	154.3	12.3	5.4	138.5	12.4			
8				4.6	135.3	12.3	5.7	162.9	12.2	5.3	135.9	12.4			
9				4.4	129.4	12.1	6.4	182.9	12.2						
10															
Averages	4.2	105.5	12.2	4.7	139.5	12.3	5.1	144.8	12.2	5.4	137.2	12.3	4.7	138.2	12.3

*Grams $Ca(OH)_2$ per kilogram total solids in the raw sludge.

No. of Process Control Checks Made	August 6			August 7			August 8			August 13			August 14		
	$Ca(OH)_2$ Dose			$Ca(OH)_2$ Dose			$Ca(OH)_2$ Dose			$Ca(OH)_2$ Dose			$Ca(OH)_2$ Dose		
	g/ℓ	g/kg*	pH	g/ℓ	g/kg*	pH	g/ℓ	g/kg*	pH	g/ℓ	g/kg*	pH	g/ℓ	g/kg*	pH
1	4.9	140.0	12.3	4.4	200.0	12.2	4.4	125.7	12.3	6.3	218.8	12.4	4.7	95.9	12.1
2	4.9	140.0	12.4	4.4	200.0	12.2	4.4	125.7	12.3	6.1	174.3	12.4	4.8	97.9	12.3
3	4.9	140.0	12.3	4.6	209.1	12.3	4.7	134.3	12.3	5.8	165.7	12.4	4.5	91.8	12.3
4	4.9	140.0	12.2	4.6	209.1	12.4	4.8	137.1	12.3	5.7	162.9	12.4	5.0	102.0	12.3
5	5.0	142.9	12.4	4.8	218.2	12.4	5.0	142.9	12.3	5.5	157.1	12.3	5.0	102.0	12.3
6	5.0	142.9	12.4	4.8	218.2	12.4	5.0	142.9	12.3	5.5	157.1	12.3	5.4	110.2	12.4
7	4.9	140.0	12.3	4.4	200.0	12.4	5.0	142.9	12.3	5.3	151.4	12.3	5.3	108.2	12.4
8	5.0	142.9	12.3							5.2	148.6	12.3	5.2	106.1	12.4
9	4.9	140.0	12.2							5.6	160.0	12.3	5.2	106.1	12.4
10	5.3	151.4	12.3							5.8	165.7	12.4			
Averages	5.0	142.0	12.3	4.6	207.8	12.3	4.8	135.9	12.3	5.7	166.2	12.4	5.0	102.2	12.3

*Grams $Ca(OH)_2$ per kilogram total solids in the raw sludge

This equation suggests that the greatest portion of the lime requirement is associated with the liquid phase and only a small fraction of the lime demand is dependent upon the solids concentration. It should be recognized, however, that the above equation describes only the initial lime demand and does not take pH decay with time into account.

Data used to derive the relationship were obtained during pilot plant operation when lime dose was adjusted to maintain a pH range between 12.2 and 12.4 and lime-sludge contact time was 30 minutes. Lime Dose is defined as the amount of lime required to satisfy the chemical demand present in the sludge and to provide the hydroxyl ion concentration necessary to raise the pH to the desired level. The total sludge chemical demand is a combination of the demand present in the liquid phase and that present in the solid phase. The demand present in the sludge liquid phase is largely governed by the reaction of the lime with dissolved CO_2 and bicarbonate ion. This demand is probably satisfied with relatively short lime-sludge contact time. The solids demand is characterized by much slower reactions of hydroxyl ions with organic materials in the sludge (neutralizing organic acids, hydrolysis, and saponification) so that this demand may be exerted over long periods of time (hours or days). This long term demand exerted by the sludge solids causes the pH decay discussed earlier and may account in part for the greater lime doses required to reach pH 12.4 in the laboratory jar tests than in the pilot plant study. In the jar tests, system pH was allowed to equilibrate after each incremental lime dose, so that several hours were usually required to reach pH 12.4. During this time period, hydroxyl ions were satisfying liquid and solids demand as well as elevating pH. In the pilot plant study, the sludge received slug doses of lime to elevate and maintain $pH \geq 12.0$ after a 30 minute sludge-lime contact time. Therefore, in the laboratory tests, more time was available for reaction with organic material in the sludge solids and thus more lime was required in the pilot plant study. In conclusion, the lime dose required to achieve pH≥12.0 is significantly affected by the chemical demand exerted by the chemical components in the sludge liquid and solid phases, and the long term chemical demand is a function of the sludge total solid concentration.

The results derived in this study also indicate that the lime dose required to maintain the pH at or above the desired level will be affected by the natural variability of sludge chemical composition and by any type of sludge treatment which alters the sludge chemical makeup. Therefore, in practice, lime dose requirements would have to be determined for each specific sludge to be treated.

During pilot operations, studies were made of the reductions in pathogenic organisms achieved by lime treatment in the pilot process. Once again the indicator organisms used were fecal coliforms, fecal streptococci, *Salmonella* species, and *Pseudomonas aeruginosa*. Typical results from these studies are shown in Table 5 and 6.

These results show that significant pathogenic bacteria reductions can be achieved in sludges which have been continuously lime treated to pH≥12.0. Reductions of fecal coliforms and fecal streptococci were consistently greater than 99 percent. Salmonella species and Pseudomonas aeruginosa appear to be almost totally inactivated by lime stabilization.

Studies to determine the effect of lime treatment on sludge settling characteristics were conducted on sludge processed in the pilot process. One liter samples of raw and lime treated sludges were placed in graduated cylinders and allowed to settle for a specified length of time. The sludge volume at the sludge-supernatant interface was read and recorded

TABLE 5

FECAL COLIFORM AND FECAL STREPTOCOCCI IN UNTREATED AND TREATED SLUDGE SAMPLES (PILOT RUNS MADE ON MIXED PRIMARY SLUDGE AND HUMUS UNLESS OTHERWISE NOTED)

Initial pH Value	Lime Contact Time (hrs)	pH when Sample Taken	Fecal Coliform per 100 ml	Fraction of Original Remaining	Fecal Streptococci per 100 ml	Fraction of Original Remaining
July 25, 1973						
Untreated Sludge						
6.2	0.0	6.2	4.95×10^7		5.00×10^6	
6.2	0.0	6.2	3.70×10^7		5.50×10^6	
Treated Sludge						
12.2	0.5	12.2	500	1.16×10^{-5}	100	1.90×10^{-5}
12.2	0.5	12.2	<1000	9.25×10^{-5}	0	0.0
July 31, 1973 (1)						
Untreated Sludge						
5.9	0.0	5.9	5.20×10^7		1.89×10^7	
5.9	0.0	5.9	5.45×10^7		1.88×10^7	
Treated Sludge						
12.0	0.5	11.7	1.00×10^4	1.87×10^{-4}	200	1.06×10^{-5}
12.0	0.5	11.7	2.50×10^4	4.69×10^{-4}	0	0.0

---

(1)Primary sludge

TABLE 5 (Cont'd.)

FECAL COLIFORM AND FECAL STREPTOCOCCI IN UNTREATED AND TREATED SLUDGE SAMPLES (PILOT RUNS MADE ON MIXED PRIMARY SLUDGE AND HUMUS UNLESS OTHERWISE NOTED)

Initial pH Value	Lime Contact Time (hrs)	pH when Sample Taken	Fecal Coliform per 100 ml	Fraction of Original Remaining	Fecal Streptococci per 100 ml	Fraction of Original Remaining
August 8, 1973 (2)						
Untreated Sludge						
6.1	0.0	6.1	4.5×10^7		9.23×10^6	
6.1	0.0	6.1	3.6×10^7		8.46×10^6	
Treated Sludge						
12.3	0.5	12.3	<1000	$<2.47 \times 10^{-5}$	560	6.33×10^{-5}
12.3	0.5	12.3	<1000	$<2.47 \times 10^{-5}$	33	3.73×10^{-6}
August 14, 1973 (3)						
Untreated Sludge						
6.2	0.0	6.2	9.25×10^7		2.24×10^7	
6.2	0.0	6.2	6.35×10^7		1.60×10^7	
Treated Sludge						
12.2	0.5	12.2	<1000	$<1.28 \times 10^{-5}$	100	5.21×10^{-6}
12.2	0.5	12.2	500	6.41×10^{-6}	170	8.85×10^{-6}

---

(2) Humus
(3) Thickened mixed sludge

TABLE 6

SALMONELLA SPECIES AND PSEUDOMONAS AERUGINOSA IN UNTREATED AND TREATED SLUDGE SAMPLES (PILOT RUNS MADE ON MIXED PRIMARY SLUDGE AND HUMUS UNLESS OTHERWISE NOTED)

Initial pH Value	Lime Contact Time (hrs)	Ph when Sample Taken	Salmonella Species MPN per 100 ml	Fraction of Original Remaining	Pseudomonas Aeruginosa MPN per 100 ml	Fraction of Original Remaining
July 25, 1973						
Untreated Sludge						
6.2	0.0	6.2	5200		34,000	
6.2	0.0	6.2	5400		22,000	
Treated Sludge						
12.2	0.5	12.2	0	0.0	0	0.0
12.2	0.5	12.2	0	0.0	0	0.0
July 31, 1973 (1)						
Untreated Sludge						
5.9	0.0	5.9	6800		<320,000	
5.9	0.0	5.9	7800		<320,000	
Treated Sludge						
12.0	0.5	11.7	0	0.0	0	0.0
12.0	0.5	11.7	0	0.0	0	0.0

---

(1) Primary sludge

TABLE 6 (Cont'd.)

SALMONELLA SPECIES AND PSEUDOMONAS AERUGINOSA IN UNTREATED AND TREATED SLUDGE SAMPLES (PILOT RUNS MADE ON MIXED PRIMARY SLUDGE AND HUMUS UNLESS OTHERWISE NOTED)

Initial pH Value	Lime Contact Time (hrs)	pH when Sample Taken	Salmonella Species MPN per 100 ml	Fraction of Original Remaining	Pseudomonas Aeruginosa MPN per 100 ml	Fraction of Original Remaining
August 8, 1973 (2)						
Untreated Sludge						
6.1	0.0	6.1	2600		22,000	
6.1	0.0	6.1	7000		15,800	
Treated Sludge						
12.3	0.5		0	0.0	0	0.0
12.3	0.5		0	0.0	0	0.0
August 14, 1973 (3)						
Untreated Sludge						
6.2	0.0	6.2	10,800		56,000	
6.2	0.0	6.2	7000		56,000	
Treated Sludge						
12.2	0.5	12.2	0	0.0	0	0.0
12.2	0.5	12.2	0	0.0	0	0.0

---

(2) Humus
(3) Thickened mixed sludge

periodically. The sludge samples were also gently stirred periodically to eliminate the effect of bridging among sludge particles. Typical results from these tests are shown in Figures 3-6.

In all but one instance, sludge settling characteristics were enhanced by lime treatment. This phenomena is probably caused by the formation of floc which settles better than the dispersed sludge particles in the raw sludge. The supernatants recovered from these tests were clear and had total solids concentrations ranging from 0.1 to 0.3 percent and 0.5 to 0.7 percent in the raw sludge and treated sludge supernatants, respectively. The higher total solids concentrations from the treated sludge supernatants are caused by the high concentrations of dissolved $Ca(OH)_2$ introduced in the lime slurry and from increases in dissolved organic concentration caused by hydrolysis and saponification reactions.

These results indicate that lime treatment of sludges prior to thickening operations would enhance the effectiveness of the thickener. Removal of a portion of the sludge liquid phase would reduce the overall volume of the sludge which must receive further treatment or removal from the treatment plant. If the thickened, lime treated sludges were to be applied to agricultural land, removal of a portion of the liquid phase would reduce the volume of sludge to be transported to the disposal site. The high pH conditions created by lime treatment would also prevent odor production in thickeners enabling the use of longer residence times which would improve the effectiveness of the thickener.

Growth Studies

The phase of the program dealing with the effects of spreading lime stabilized sludges on land used for crop production involved both greenhouse studies and larger scale outdoor plot studies.

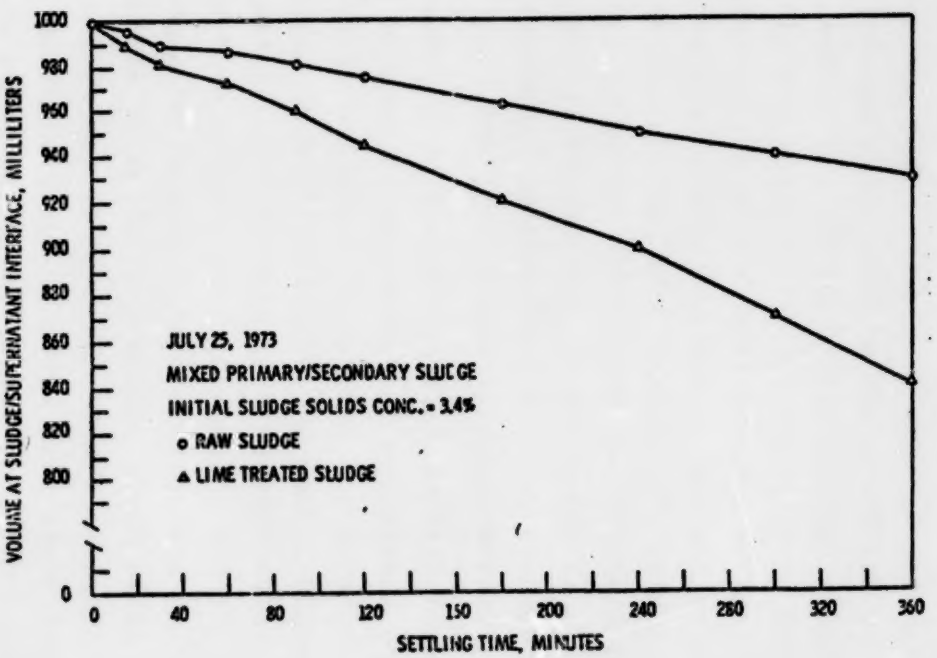

FIGURE 3. EFFECT OF LIME TREATMENT ON SLUDGE SETTLING CHARACTERISTICS - 6/25/73

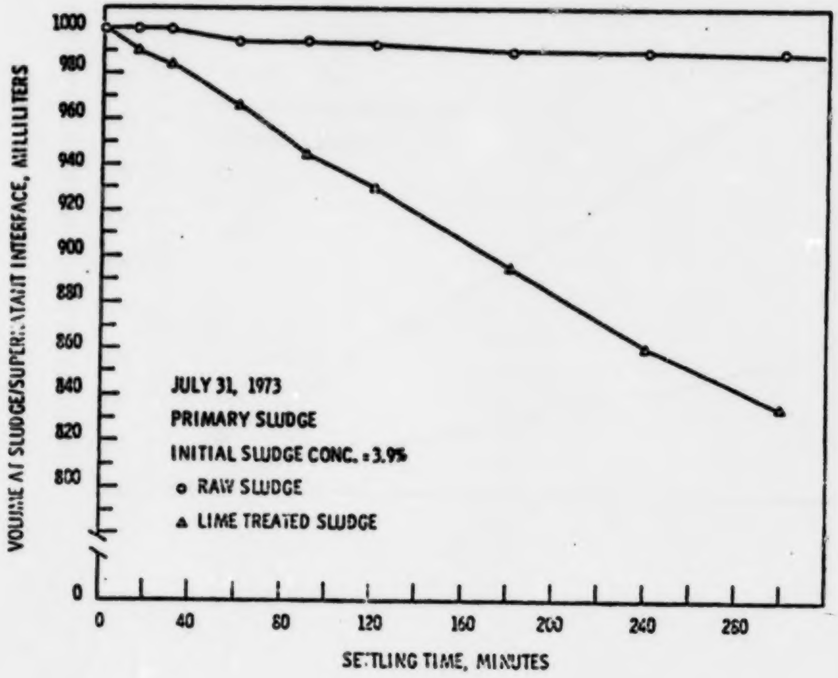

FIGURE 4. EFFECT OF LIME TREATMENT ON SLUDGE SETTLING CHARACTERISTICS - 6/31/73

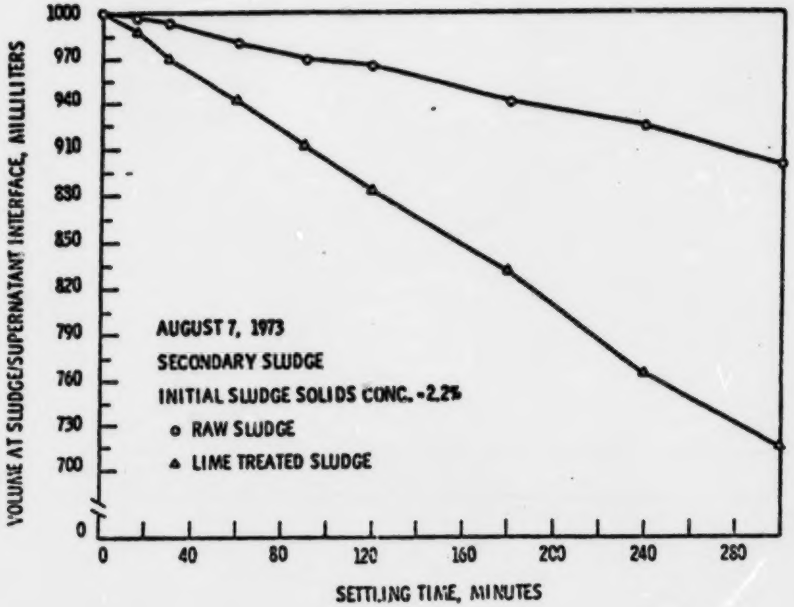

FIGURE 5. EFFECT OF LIME TREATMENT ON SLUDGE SETTLING CHARACTERISTICS - 7/7/73

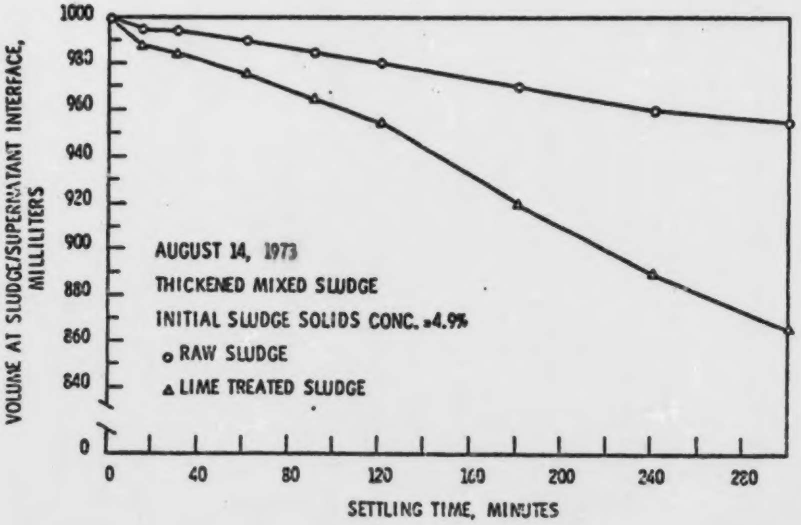

FIGURE 6. EFFECT OF LIME TREATMENT ON SLUDGE SETTLING CHARACTERISTICS - 7/14/73

The greenhouse studies were designed to provide information on the response of plants grown in various sludge-soil mixtures. These studies also yielded information which was used in design of the outdoor plot studies. The outdoor plot studies were conducted during the summer of 1973 at the Washington State University Irrigated Agriculture Research and Extension Center in Prosser, Washington.

For the greenhouse studies, anaerobically digested and lime stabilized raw sludges in liquid form were applied to small outdoor plots at five application rates ranging from 11 to 220 metric tons per hectare (5 to 100 tons per acre). The sludges dewatered by the mechanisms of draining and evaporation, and the sludge solids were left on the surface of the plots. After the sludge dried, the solids were spaded into the underlying soil to approximately plow depth, 20 cm (8 inches). Sludge-soil mixtures from each were placed in pots and barley growth was carried out. This technique for sludge application had the advantage of very closely simulating conditions encountered in large scale sludge spreading operations.

The sludge-soil mixtures were placed in clay flower pots and readied for use. A set of control pots was prepared for use in comparing plant growth characteristics and soil response to sludge application. The control set, which contained only soil with no sludge additions, received optimum additions of chemical fertilizer during the actual plant growth phase of the studies. Barley was sown in the pots and the growing plants maintained through a full growth cycle as indicated by the formation of grain heads.

After the full growth cycle, the plant material and sludge-soil mixtures were subjected to analyses. The plant tissue was weighed to determine the mass yield and then chemically analyzed for available micro- and macronutrient content. The sludge-soil mixtures were analyzed both before and after plant growth for available micro- and macronutirent content, pH, permability with water, hydraulic conductivity, and field capacity.

The results from analyses of the physical characteristics of the sludge-soil mixtures used in the greenhouse studies are shown in Table 7. The soil used for mixture with sludge was classified as a Rupert sand which is very porous. Addition of sludge to the soil appears to have reduced the soil's permeability with water. This would be expected in a sandy soil since the sludge organic matter acts to retain moisture. No well defined trend developed which would correlate permeability with the amount of sludge applied. In general, permeability appears to be reduced after plant growth. This could possibly be caused by biodegradation of the coarse organic components in the sludge to finer humus-like material which would fill pore spaces between larger sand particles.

The pH values in the sludge-soil mixture were usually lower after plant growth than before. This phenomenon is believed to be caused by CO_2 buildups resulting from biological activity in the soil.

The field capacities of the sludge-soil mixtures were observed to be lower in the samples taken after plant growth than in those taken before the barley was planted.

The results obtained from analyses of sludge-soil mixtures for available macro- and micronutrients before and after the plant growth are shown in Table 8. In general, the results show increases in available nutrient concentrations in the sludge-soil mixtures as sludge application rates increased. A decrease in available nutrient concentrations apparently occurs during plant growth. This decrease is probably caused by nutrient uptake in the growing plants. Sludge application to the soil at rates as low as 11 metric tons dry solids per

TABLE 7

PHYSICAL CHARACTERISTICS OF SOILS BEFORE AND AFTER BARLEY GROWTH IN THE GREENHOUSE STUDY

Sludge/Soil Type and Sludge Application Rate (tons dry solids/acre)	Intrinsic Permeability with Water - K'_w (cm^2)		Hydraulic Conductivity - K (cm/sec)		K_m/K_c [1]		pH of Mixture[2]		Field Capacity of Mixture[3] (% of soil dry wt.)	
	Pre-Gro	Post-Gro	Pre-Gro	Post-Gro	Pre-Gro	Post-Gro	Pre-Gro	Post-Gro	Pre-Gro	Post-Gro
Control										
100% Rupert Sand (RS)	3.91×10^{-8}	1.44×10^{-8}	4.61×10^{-3}	1.74×10^{-3}	1.00	0.38	7.6	6.1	5.5	4.6
Mixed Primary Humus and RS										
5	3.36×10^{-8}	9.91×10^{-8}	3.96×10^{-3}	1.17×10^{-3}	0.86	0.25	7.8	7.6	9.1	7.1
30	1.91×10^{-8}	1.70×10^{-8}	2.25×10^{-3}	2.00×10^{-3}	0.49	0.43	8.1	8.1	9.4	8.8
55	3.67×10^{-8}	1.84×10^{-8}	4.34×10^{-3}	2.17×10^{-3}	0.94	0.47	8.5	8.1	13.9	10.2
80	2.33×10^{-8}	1.73×10^{-8}	2.75×10^{-3}	2.04×10^{-3}	0.60	0.44	10.6	8.1	15.9	9.6
100	4.82×10^{-8}	2.85×10^{-8}	5.69×10^{-3}	3.36×10^{-3}	1.23	0.72	10.2	8.2	11.4	5.3
Digested Sludge and RS										
5	2.45×10^{-8}	1.31×10^{-8}	2.90×10^{-3}	1.54×10^{-3}	0.63	0.33	7.0	7.3	8.6	6.8
30	1.51×10^{-8}	2.58×10^{-8}	1.78×10^{-3}	3.04×10^{-3}	0.39	0.66	6.5	6.8	13.3	8.2
55	1.57×10^{-8}	3.50×10^{-8}	1.85×10^{-3}	4.13×10^{-3}	0.40	0.89	6.7	6.6	15.2	12.4
80	1.91×10^{-8}	2.53×10^{-8}	2.25×10^{-3}	2.99×10^{-3}	0.49	0.64	6.8	6.3	15.6	11.1
100	3.97×10^{-8}	3.88×10^{-8}	4.69×10^{-3}	4.58×10^{-3}	1.01	0.99	6.8	6.7	16.1	9.2
Primary Sludge and RS										
5	2.05×10^{-8}	6.86×10^{-9}	2.42×10^{-3}	8.09×10^{-4}	0.53	0.18	7.8	7.6	7.6	4.2
30	2.46×10^{-8}	1.06×10^{-8}	2.90×10^{-3}	1.25×10^{-3}	0.63	0.27	7.6	7.6	8.4	5.4
55	2.77×10^{-8}	2.58×10^{-8}	3.27×10^{-3}	3.04×10^{-3}	0.71	0.66	7.9	7.6	10.7	4.2
80	4.30×10^{-8}	3.14×10^{-8}	5.07×10^{-3}	3.70×10^{-3}	1.10	0.80	8.1	7.7	14.9	6.4
100	2.68×10^{-8}	3.26×10^{-8}	3.16×10^{-3}	3.85×10^{-3}	0.69	0.84	10.0	7.7	15.0	5.5
Humus and RS										
5	2.60×10^{-8}	1.22×10^{-8}	3.07×10^{-3}	1.44×10^{-3}	0.66	0.31	7.9	8.2	7.7	3.5
30	2.32×10^{-8}	2.02×10^{-8}	2.74×10^{-3}	3.39×10^{-3}	0.59	0.74	8.5	8.3	13.2	5.6
55	5.09×10^{-8}	5.38×10^{-8}	6.01×10^{-3}	6.36×10^{-3}	1.30	1.38	8.0	8.0	13.2	4.4
80	4.42×10^{-8}	4.35×10^{-8}	5.22×10^{-3}	5.14×10^{-3}	1.13	1.11	8.1	8.0	16.1	4.4
100	2.22×10^{-8}	4.26×10^{-8}	2.62×10^{-3}	5.04×10^{-3}	0.56	1.09	8.3	8.0	20.2	6.4

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1. Ratio of the hydraulic conductivity of the sludge-soil mixture (K_m) to that of the control (K_c) before the growth cycle.
2. Soil pH measured in water.
3. 1/3 bar percentage.

TABLE 8

MACRO- AND MICRONUTRIENT CONCENTRATIONS IN SLUDGE-SOIL MIXTURES BEFORE AND AFTER BARLEY GROWTH IN THE GREENHOUSE STUDY

Sludge/Soil Type and Sludge Application Rate (tons dry solids/acre)	Nitrate-N (ppm)		Phosphorus (ppm)		Potassium (ppm)		Sulfur (ppm)		Magnesium (ppm)		Calcium (ppm)	
	Pre-Gro	Post-Gro	Pre-Gro	Post-Gro	Pre-Gro	Post-Gro	Pre-Gro	Post-Gro	Pre-Gro	Post-Gro	Pre-Gro	Post-Gro
Control												
100% Rupert Sand (RS)	2	37	7	100	280	106	1	10	108	144	360	320
Mixed Primary and Secondary Sludge and RS												
5	13	3	9	13	200	150	9	5	120	120	520	720
30	43	2	88	41	210	110	50	26	132	96	2160	1900
55	25	2	125	60	190	120	48	33	120	108	2200	2400
80	2	2	175	78	210	150	0	0	120	132	3040	2800
100	2	3	275	50	240	120	0	24	132	132	3080	2300
Digested Sludge and RS												
5	19	2	18	11	240	150	44	12	120	120	460	420
30	35	1	40	56	240	80	140	23	132	96	540	520
55	3	7	85	104	240	100	150	120	120	96	560	560
80	1	23	46	110	190	80	200	185	120	120	680	600
100	1	4	116	122	190	80	210	120	132	96	720	600
Primary Sludge and RS												
5	11	1	16	20	170	120	0	3	120	120	600	460
30	36	14	60	104	210	110	46	25	120	120	1660	920
55	34	2	125	69	240	100	57	16	132	96	2100	1320
80	1	1	225	130	240	110	0	27	120	96	2760	1800
100	2	2	220	110	240	120	0	48	120	120	2640	2040
Humus and RS												
5	19	2	15	16	200	140	12	4	120	120	520	560
30	4	3	60	59	190	110	12	4	120	96	1120	1600
55	15	2	150	130	210	90	39	9	108	132	1460	1460
80	1	2	200	244	290	80	60	30	144	132	1660	2200
100	2	26	250	244	290	100	0	44	168	144	2000	2080

TABLE 8 (Cont'd.)

MACRO- AND MICRONUTRIENT CONCENTRATIONS IN SLUDGE-SOIL MIXTURES BEFORE AND AFTER BARLEY GROWTH IN THE GREENHOUSE STUDY

Sludge/Soil Type and Sludge Application Rate (tons dry solids/acre)	Sodium (ppm)		Iron (ppm)		Manganese (ppm)		Boron (ppm)		Copper (ppm)		Zinc (ppm)	
	Pre-Gro	Post-Gro	Pre-Gro	Post-Gro	Pre-Gro	Post-Gro	Pre-Gro	Post-Gro	Pre-Gro	Post-Gro	Pre-Gro	Post-Gro
Control												
100% Rupert Sand (RS)	75	33	68	73	37	26	0.5	0.5	9	5	1	4
Mixed Primary and Secondary Sludge and RS												
5	50	44	197	42	95	102	0.4	0.6	8	6	2	7
30	87	55	210	106	70	63	0.6	0.7	10	7	27	21
55	110	67	217	117	180	140	0.6	1.0	9	12	26	26
80	112	112	245	103	85	70	0.5	0.8	11	10	50	37
100	142	105	177	208	137	133	0.5	0.5	12	9	58	26
Digested Sludge and RS												
5	27	23	288	56	84	52	0.8	0.6	7	10	8	15
30	80	95	310	60	90	86	1.1	0.7	11	12	46	38
55	90	85	211	187	76	102	1.4	1.1	10	6	68	88
90	145	77	214	210	66	140	1.7	1.3	9	8	33	125
100	160	93	186	200	83	92	1.3	1.5	13	12	172	105
Primary Sludge and RS												
5	52	67	53	47	40	105	0.8	0.4	11	6	3	7
30	30	63	70	57	65	88	0.9	1.0	9	11	30	27
55	140	85	93	100	93	115	0.9	0.5	12	7	26	20
80	76	112	112	60	130	105	0.8	0.9	10	9	70	30
100	126	107	312	94	100	95	0.5	0.6	14	12	48	40
Humus and RS												
5	28	43	108	85	85	110	0.4	0.5	9	8	2	5
30	33	40	88	96	97	62	0.7	1.2	12	7	10	10
55	122	113	96	105	83	105	0.9	0.6	11	12	22	18
80	207	86	300	80	70	72	1.0	0.8	10	12	40	36
100	237	120	420	106	100	55	1.1	0.9	12	9	37	34

hectare (5 tons dry solids per acre) significantly increased the concentrations of available calcium and iron in the mixtures. The increase in calcium concentration was expected because of the lime added to the sludges. Application of moderate to high amounts of sludge caused significant increases in the concentrations of available phosphorus, sodium, manganese, and zinc.

Results from the study of barley weight gains are shown in Table 9. The control pots which received only chemical fertilizer yielded plants which averaged only 4.7 grams each. However, the addition of sludge to the soils significantly affected the yield. The sludge-soil mixtures made from mixed primary sludge and humus, primary sludge alone, and humus alone, applied at the lowest rate of 11 metric tons dry solids per hectare (5 tons dry solids per acre) all yielded less plant material than the control pots which received only chemical fertilizer. The mixture made from Rupert sand and digested sludge applied at 11 metric tons dry solids per hectare (5 tons dry solids per acre) produced plants whose average weight exceeded that of the control by almost 2.5 times. The mixtures made from mixed sludge and digested sludge applied to Rupert sand all produced increasing plant material yields as the sludge application rates increased from 66 through 176 metric tons dry solids per hectare (30 through 80 tons dry solids per acre). Plant yield decreased for each of these sludge-soil types when the application rate reached 220 metric tons dry solids per hectare (100 tons dry solids per acre). The mixtures made from primary sludge and humus applied to Rupert sand produced increasing plant yields as sludge application rates increased through 220 metric tons per hectare (100 tons per acre).

These results indicate that sludge addition to poor soils would increase productivity and therefore would be beneficial. The addition of large amounts of lime to the sludges did not appear to produce any detrimental effects.

TABLE 9

BARLEY WEIGHT GAINS FROM THE GREENHOUSE STUDY

Sludge/Soil Type and Sludge Application Rate (tons dry solids/acre)	Total Number Plants	Total Weight of All Plant Tissue Produced (grams)	Average Weight of Tissue in Each Plant (grams/plant)	Yield Ratio* (grams/gram)
Control				
100% Rupert Sand (RS)	16	74.8	4.7	1.00
Mixed Primary Sludge and Humus and RS				
5	16	67.0	4.2	0.89
30	16	158.6	9.9	2.12
55	15	200.1	13.3	2.83
80	15	252.6	16.8	3.57
100	16	206.5	12.9	2.74
Digested Sludge and RS				
5	14	162.1	11.6	2.47
30	15	211.6	14.1	3.00
55	17	253.2	14.9	3.17
80	16	309.4	19.4	4.13
100	15	219.2	14.6	3.11
Primary Sludge and RS				
5	16	36.1	2.3	0.49
30	14	122.8	8.8	1.87
55	16	144.5	9.0	1.91
80	17	122.2	7.2	1.53
100	17	176.4	10.4	2.21
Humus and RS				
5	15	41.2	2.8	0.60
30	16	122.2	7.6	1.62
55	16	175.6	11.0	2.34
80	14	238.6	17.0	3.62
100	16	297.2	18.6	3.96

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*Calculated as grams plant tissue from the sludge-soil mixtures per gram plant tissue from the control pots.

Results from analysis of macro- and micronutrient content of the barley grown in this study are shown in Table 10. The total nitrogen and phosphorus levels in the plants grown in the test pots which contained sludge-soil mixtures were consistently lower than in the plants grown in the control which contained only soil. These results cannot be interpreted as indicating a nitrogen or phosphorus deficiency in the soils which received sludge treatment since plant production in these pots generally exceeded that in the control pots. The calcium concentration in plant tissues from pots which received sludge applications was higher than in the plant tissue from the control pots. Zinc concentration was considerably higher in the tissue of plants grown in pots which received digested sludge than in any of the other plant tissues tested.

In order to further evaluate the short term effects of spreading lime treated sludge on crop land, larger scale crop growth studies were conducted on outdoor plots at the Washington State University Irrigated Agriculture Experiment and Extension Center in Prosser, Washington. The soil at the site was classified as Warden silt loam. The site had not been used for any agricultural experiments during the preceding year. Five 0.04 hectar (0.1 acre) plots were used: one control plot received no sludge (only application of optimum fertilizer requirements); two plots received applications of anaerobically digested sludges at rates equivalent to 22 and 88 metric tons dry solids per hectare (10 and 40 tons per acre); and two plots received lime treated mixed primary sludge and humus at the same application rates used for the digested sludge. Buffer zones were provided between plots to assure individual plot integrity during sludge spreading operations, plant growth, and harvesting operations. The sludge was transported to the site by a contracted septic tank service man. Even distribution of the sludge on the plots was accomplished by use of a splasher plate attached to the tank truck discharge port.

TABLE 10

MACRO- AND MICRONUTRIENTS IN BARLEY TISSUE FROM THE GREENHOUSE STUDY (ALL SLUDGES LIME TREATED EXCEPT THE DIGESTED SLUDGE)

Sludge/Soil Type and Sludge Application Rate (tons dry solids/acre)	Total N (%)	Phosphorus (%)	Potassium (%)	Sulfur (%)	Magnesium (%)	Calcium (%)	Sodium (%)	Iron (ppm)	Manganese (ppm)	Boron (ppm)	Copper (ppm)	Zinc (ppm)
Control												
100% Rupert Sand (RS)	4.1	1.14	3.10	0.32	0.98	0.75	0.58	478	36	31	11	31
Mixed Primary and Secondary Sludge and RS												
5	1.2	0.45	3.48	0.32	0.37	1.12	0.21	450	68	14	11	70
30	1.3	0.62	4.32	0.49	0.44	1.23	0.26	700	73	16	16	107
55	1.6	0.51	3.60	0.38	0.45	1.20	0.38	800	98	16	14	93
80	1.5	0.46	3.23	0.34	0.45	1.45	0.63	433	122	24	11	68
100	1.9	0.52	3.41	0.43	0.44	1.25	0.58	445	120	21	12	74
Digested Sludge and RS												
5	1.9	0.42	4.40	0.40	0.37	0.92	0.59	800	60	23	14	92
30	2.6	0.56	3.40	0.40	0.45	1.50	0.75	820	102	34	19	450
55	3.2	0.52	2.82	0.41	0.45	1.68	0.95	550	215	55	21	500
80	3.9	0.59	2.55	0.36	0.41	1.68	1.20	950	221	50	23	550
100	3.2	0.51	2.38	0.39	0.38	1.50	1.00	550	187	45	21	500
Primary Sludge and RS												
5	1.4	0.44	4.00	0.35	0.41	1.15	0.38	1250	55	16	13	70
30	2.1	0.39	3.75	0.39	0.41	1.20	0.70	428	44	19	13	77
55	2.1	0.39	3.41	0.48	0.37	1.37	0.85	475	60	14	11	84
80	2.0	0.55	3.87	0.50	0.37	1.12	0.70	415	53	13	14	110
100	2.7	0.42	3.05	0.49	0.38	1.25	1.00	500	65	15	14	90
Humus and RS												
5	1.2	0.50	4.00	0.31	0.38	0.87	0.38	650	44	20	9	57
30	2.6	0.45	3.60	0.36	0.37	1.20	0.75	950	97	27	15	62
55	2.5	0.70	3.55	0.40	0.40	1.55	0.85	550	145	31	15	84
80	3.0	0.54	3.23	0.40	0.43	1.55	0.95	600	138	25	16	95
100	3.2	0.42	3.00	0.40	0.43	1.75	1.00	700	135	21	16	79

After sludge application, the plots were allowed to dry and were then prepared for planting. Samples for analyses of soil physical and chemical characteristics were taken at this time. Sudan grass, an annual pasture grass adapted to Eastern Washington State, was used as an indicator plant. Maintenance of the plots during plant growth mainly involved periodic application of irrigation water and was carried out by the staff of the WSU Experiment Center. In early autumn when danger from frost damage was imminent, the grass was harvested. The yield from each plot was recorded, and plant matter and soil samples were collected for analyses. These samples were subjected to the same tests as conducted on the plant tissue and soil samples from the greenhouse studies.

The results from analyses of physical characteristics of soils before and after Sudan grass cultivation are shown in Table 11. Intrinsic permeability with water slightly improved in the soils from all plots during the growth study. The greatest improvements occurred in the plots which received sludge application of 88 metric tons dry solids per hectare.

A slight decrease in soil pH was observed in the samples collected after the Sudan grass was harvested.

Field capacity varied slightly between the beginning and the end of the growth study. No general trend could be established and the variations in most cases were not significant.

The results from analysis of macro- and micronutrients in the outdoor plots before and after plant growth are shown in Table 12. Increases in the nutrient concentration resulted from the application of sludges at the rate of 88 metric tons dry solids per hectare.

TABLE 11

PHYSICAL CHARACTERISTICS OF SOILS BEFORE AND AFTER SUDAN GRASS CULTIVATION IN THE OUTDOOR PLOT STUDIES

Sludge/Soil Type and Sludge Application Rate (tons dry solids/acre)	Intrinsic Permeability with Water - K'_w (cm^2)		Hydraulic Conductivity - K (cm/sec)		K_m/K_c[3]		pH of Mixture[4]		Field Capacity of Mixture[5] (% of soil dry wt.)	
	Pre-Gro	Post-Gro	Pre-Gro	Post-Gro	Pre-Gro	Post-Gro	Pre-Gro	Post-Gro	Pre-Gro	Post-Gro
Control[1] 100% Warden Silt Loam(WSL)	7.25×10^{-9}	7.98×10^{-9}	8.36×10^{-4}	9.42×10^{-4}	1.0	1.1	6.7	6.5	22	23
Digested and WSL 10	8.70×10^{-9}	9.43×10^{-9}	1.02×10^{-3}	1.11×10^{-3}	1.2	1.3	6.6	6.3	25	24
Lime Treated[2] and WSL 10	1.81×10^{-8}	1.96×10^{-8}	2.14×10^{-3}	2.31×10^{-3}	2.5	2.7	7.3	6.8	28	30
Digested and WSL 40	4.57×10^{-8}	5.22×10^{-8}	5.39×10^{-3}	6.16×10^{-3}	6.3	7.2	6.6	6.7	28	26
Lime Treated[2] 40	5.95×10^{-8}	6.60×10^{-8}	7.02×10^{-3}	7.79×10^{-3}	8.2	9.1	8.0	6.8	32	34

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1. Control plot received recommended chemical fertilizer application instead of sludge.
2. Lime treated sludge was a mixture of primary sludge and trickling filter humus.
3. Ratio of the hydraulic conductivity of the sludge/soil mixture (K_m) to that of the control (K_c) before the growth cycle.
4. Soil pH measured in water.
5. 1/3 bar percentage.

TABLE 12

MACRO- AND MICRONUTRIENT CONCENTRATIONS IN OUTDOOR PLOTS BEFORE AND AFTER THE OUTDOOR GROWTH STUDY

Sludge Type and Application Rate (tons dry solids/acre)		Nitrate-N (ppm)		Phosphorus (ppm)		Potassium (ppm)		Sulfur (ppm)		Magnesium (ppm)		Calcium (ppm)	
		Pre-Gro	Post-Gro	Pre-Gro	Post-Gro	Pre-Gro	Post-Gro	Pre-Gro	Post-Gro	Pre-Gro	Post-Gro	Pre-Gro	Post-Gro
Control[A]		2	12	26	29	340	300	5	6	216	168	1320	960
Digested	10	25	19	33	32	360	220	15	10	216	180	1280	960
Lime Treated[B]	10	84	6	43	26	280	180	22	7	228	180	1600	1140
Digested	40	63	7	56	132	460	270	58	16	252	192	1600	1240
Lime Treated[B]	40	87	15	104	180	460	300	54	17	216	180	2160	1560

Sludge Type and Application Rate (tons dry solids/acre)		Sodium (ppm)		Iron (ppm)		Manganese (ppm)		Boron (ppm)		Copper (ppm)		Zinc (ppm)	
		Pre-Gro	Post-Gro	Pre-Gro	Post-Gro	Pre-Gro	Post-Gro	Pre-Gro	Post-Gro	Pre-Gro	Post-Gro	Pre-Gro	Post-Gro
Control[A]		66	33	21	40	9	16	0.3	0.5	10.0	1.1	18.5	4.2
Digested	10	66	40	28	52	13	49	0.4	0.5	1.5	2.5	11.5	16.5
Lime Treated[B]	10	88	33	32	54	14	82	0.4	0.4	1.7	2.6	12.0	14.8
Digested	40	110	40	44	73	27	93	0.6	0.7	2.9	8.8	30.0	68.0
Lime Treated[B]	40	110	40	45	70	23	110	0.5	0.7	1.5	6.0	15.0	50.0

[A] Control plot received optimum chemical fertilizer application instead of sludge.

[B] Lime treated sludge was a mixture of primary and secondary sludge.

The average maximum plant heights and the green tonnage yields resulting from growth of the Sudan grass are summarized below.

Test Plot	HEIGHT cm	HEIGHT in	YIELD m. tons/ hectare	YIELD tons/ acre
Control (no sludge)	66	26	11.77	5.35
Digested Sludge				
22 metric tons dry solids/hectare	117	46	20.35	9.25
88 metric tons dry solids-hectare	132	52	24.20	11.00
Lime Stabilized Sludge				
22 metric tons dry solids/hectare	89	35	17.16	7.80
88 metric tons dry solids/hectare	132	52	25.96	11.80

The Sudan grass growth on the plots which received sludge applications was more luxurient than on the control plot which received only an optimum application of chemical fertilizer. For the plots which received 22 metric tons dry solids per hectare, the grass reached heights of 117 cm (46 inches) with digested sludge applied and 89 cm (35 inches) with lime stabilized sludge applied. The grass which received lime treated sludge had a yellowish tinge while the grass in the digested sludge plot had a healthy dark green appearance. In each of the plots which received 88 metric tons dry solids per hectare of digested and lime stabilized sludge, the grass grew to a height of 132 cm (52 inches). The plants in both of these plots exhibited dark green, healthy appearances.

Results from analysis of Sudan grass for macro- and micronutrient content are shown in Table 13. These results indicate that the amount of nutrients concentrated in the plant tissue

TABLE 13

MACRO- AND MICRONUTRIENT CONCENTRATIONS IN SUDAN GRASS TISSUE FROM OUTDOOR GROWTH STUDY

Sludge Type & Application Rate (tons dry solids/acre)		Nitrate Nitrogen ppm	P Phosphorus %	K Potassium %	S Sulphur %	Mg Magnesium %	Ca Calcium %	Na Sodium %	Fe Iron ppm	Mn Manganese ppm	B Boron ppm	Cu Copper ppm	Zn Zinc ppm
Control[1]		800	.5	3.05	.24	.40	.52	.06	370	48	7.0	10.5	66
Digested	10	100	.47	3.20	.19	.41	.77	.09	530	34	7.0	11.0	80
Lime Treated[2]	10	0	.52	2.85	.14	.42	.80	.22	1800	55	9.5	8.5	50
Digested	40	600	.48	2.75	.15	.44	1.00	.10	640	40	8.0	12.0	50
Lime Treated[2]	40	500	.42	2.37	.16	.42	.95	.09	1850	38	7.5	6.5	34

1. Control plot received recommended chemical fertilizer application instead of sludge.
2. Lime treated sludge was a mixture of primary and secondary sludge.

is independent of the amount or type of sludge applied to the land in which the plants are grown. Also, there were no indications of buildup of excessive amounts of nutrients in the plant material with the exception of calcium and iron which did show concentration increases over those in the chemically fertilized control plot.

DESIGN AND COST CONSIDERATIONS

Process Design

Based upon this work, it appears that the two most important process variables which must be considered are pH and contact time. Results indicate that the lime dose to the raw sludge should be sufficiently high to initially attain $pH \geq 12.0$. Moreover, the lime dose should be high enough to prevent significant pH decay during storage. In the laboratory and pilot plant work conducted in this program, short term (one hour) lime-sludge contact at $pH \geq 12.0$ provided excellent reduction in viable pathogenic bacteria populations, but upon storage pH was subject to decay. Therefore, in practice, excess lime should be added to maintain the desired pH level during storage.

The lime dose required to achieve and maintain high pH levels will vary considerably among different types of sludges; and even for the sludge produced at a specific treatment plant, the required dose will probably be subject to temporal variations. The quantity of lime required to achieve the desired condition in any particular sludge can be determined easily in the laboratory. Sludge samples of a known volume can be titrated with a lime slurry until the desired pH level is achieved. Sludge samples dosed with the minimum lime addition required to reach the desired pH and others dosed with increasing multiples of this amount could then be stored and pH decay observed over a period of time.

By using this procedure, a good indication of the lime dose required to achieve and maintain the desired conditions could be obtained. A full scale process can be designed with automatic process control equipment.

A possible process flow scheme is shown in Figure 7. The process flow is basically the same as that used in the pilot plant operated for this study. The main variations are provisions for automatic process control and the capability for adding excess lime in the sludge-lime contactor.

Process operation consists of introducing sludge into a mixing vessel where lime slurry is added. The pH level of the sludge in this vessel is continuously monitored and lime slurry addition automatically altered when the pH deviates from the setpoint. Sludge whose pH had been elevated to the desired level is continuously passed from the mixing vessel to a sludge-lime contactor. This contactor is also mixed and the excess lime required to prevent pH decay is added at this point. The quantity of excess lime is a specified multiple of the dose being added in the sludge-lime mixing vessel. This lime feed system provides positive process control since lime additions in the mixing vessel vary in accordance with temporal variations in sludge chemical demand. The addition of excess lime to maintain desired conditions is directly tied to the lime dose added in the mixing vessel.

Process Costs

Cost estimates for a lime stabilization process must be based on laboratory and pilot plant information. Lime costs may be easily and accurately estimated from chemical dose data from laboratory and pilot plant work. The chemical cost estimates made in this section were based on a hydrated lime cost of \$22 per metric ton (\$22 per short ton). Operating and maintenance (O&M) costs were estimated as shown below.

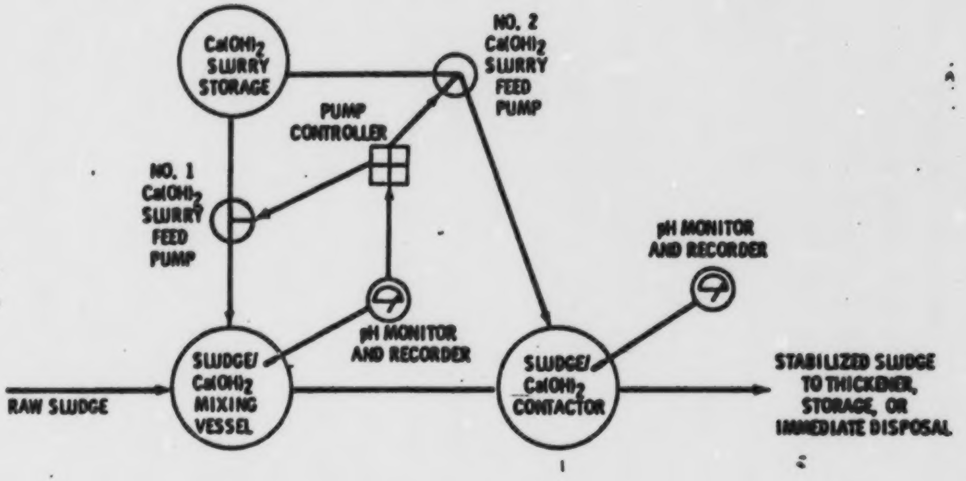
Ca(OH)$_2$ SLURRY STORAGE
NO. 2 Ca(OH)$_2$ SLURRY FEED PUMP
PUMP CONTROLLER
NO. 1 Ca(OH)$_2$ SLURRY FEED PUMP
pH MONITOR AND RECORDER
pH MONITOR AND RECORDER
RAW SLUDGE
SLUDGE/ Ca(OH)$_2$ MIXING VESSEL
SLUDGE/ Ca(OH)$_2$ CONTACTOR
STABILIZED SLUDGE TO THICKENER, STORAGE, OR IMMEDIATE DISPOSAL

FIGURE 7. LIME STABILIZATION PROCESS CONCEPTUAL FLOWSHEET

	COSTS ($/metric ton sludge solids)
Electricity	$0.76
Operating Labor	4.32
Maintenance Labor	1.22
Repair Materials	0.44
Other Operating Costs (not included above)	0.26
Total Estimated O&M Costs	$7.00

The $0.26 included under "Total Operating Costs" was added to account for sludge pumping and mixing costs which would have been excluded otherwise. For a 37,850 m^3/day (10 MGD) sewage treatment plant which produces a total sludge flow of approximately 255 m^3/day (67,000 gal/day), the total capital cost of a lime stabilization process would probably be less than $8000. This cost includes tankage, piping, chemical feed system, and automatic control instrumentation. This cost is too small to be financed by a bond issue and would probably be paid directly from an account set to finance such low cost improvements of municipal facilities. Since capital costs are considered insignificant, the major cost of the lime stabilization process would be O&M costs which, as stated above, would amount to approximately $7.00 per metric ton of sludge solids treated.

From his work in Ohio, Farrell[1] estimated that lime addition to an alum primary sludge cost an average of $4.95 per metric ton sludge solids. By adding the O&M costs developed previously to this chemical cost, a total O&M cost estimate of $11.95 per metric ton sludge solids was obtained. Farrell[1] also found that iron primary sludges had an average lime cost of $2.50 per metric ton sludge solids. Therefore, total O&M costs in this case would be about $9.50 per metric ton sludge solids. The amount of lime applied to the sludges in this study was, in general, the minimum dose required to raise pH to 11.5. Excess lime to maintain pH above a specified level was not added.

O&M cost estimates based on the work done by Paulsrud and Eikum[3] were also developed for comparison purposes. The dose required and the estimated costs for lime stabilization of various types of sludges are summarized in Table 14. The recommended lime doses are those required to maintain $pH \geq 11.0$ in sludges stored for fourteen days at 20°C. The total estimated O&M costs using these recommended lime doses range from $9 to $19 per metric ton sludge solids. These results indicate that treatment costs will be mainly dependent on chemical requirements which will vary with the type of sludge being treated and the chemical pretreatment history of the sludge.

Process costs developed from the results of this program agree with those developed from the results of the other investigators. Chemical cost estimates were based on an average lime dose of 150 g $Ca(OH)_2$/kg sludge total solids required to achieve $pH \geq 12.0$ and maintain that level for one hour. Lime costs in this case were found to be $3 per metric ton sludge solids, so that estimated total O&M costs would be $10 per metric ton sludge solids.

A comparison of approximate capital and operating costs for various sludge stabilization processes is shown in Table 15.

PROCESS APPLICATIONS

There are several situations where application of the lime stabilization process could be advantageous. Small treatment plants which do not produce large quantities of sludge and have access to land for disposal by spreading could certainly use a simple, reliable, and inexpensive sludge treatment process. Another possible strategy, as suggested by

TABLE 14

O&M COSTS FOR LIME STABILIZATION

Type of Sludge	(g $Ca(OH)_2$/kg SS)	Lime Costs ($/metric ton)	Total[2] O&M Costs ($/per metric ton)
Primary sludge	100 - 150	2.00 - 3.00	9.00 - 10.00
Septic tank sludge	100 - 300	2.00 - 6.00	9.00 - 13.00
Biological sludge	300 - 500	6.00 - 10.00	13.00 - 17.00
Al-sludge (Secondary precipitation)	400 - 600	8.00 - 12.00	15.00 - 19.00
Al-sludge (Secondary precipitation + Prim. sludge (SS_{Al}:SS_{PRIM} = 1:1)[1]	250 - 400	5.00 - 8.00	12.00 - 15.00
Fe-sludge (Secondary precipitation)	350 - 600	7.00 - 12.00	14.00 - 19.00

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[1] SS = suspended solids.

[2] Sum of lime cost and $7.00 O&M cost estimated previously.

TABLE 15

APPROXIMATE CAPITAL AND OPERATING COSTS FOR VARIOUS SLUDGE STABILIZATION PROCESSES

	Anaerobic Digestion $ per Metric Ton Dry Solids	Multiple Hearth Incineration $ per Metric Ton Dry Solids	Lime Stabilization $ per Metric Ton Dry Solids
O&M Cost	7.00	12.00	10.00
Capital Cost	24.00	23.00	Insignificant
Total Cost	31.00	35.00	10.00

Paulsrud and Eikum,[3] is for small treatment plants to use lime stabilization as a preparatory step for sludge storage. The stored, lime treated sludge would be periodically hauled away to larger facilities for further treatment and/or disposal.

For plants which utilize digestion and do not have excess digester capacity, lime treatment may provide a satisfactory means of stabilizing sludge prior to ultimate disposal. Sludge flows in excess of digester capacity could be bypassed to a separate lime treatment facility. Another option would be to use existing digesters to thicken lime treated sludge prior to dewatering or disposal.

Lime stabilization could also be used as a stop-gap technique when digesters or other sludge treatment processes temporarily are not working. In this context, lime stabilization would be used as an emergency backup process. In a temporary process, sludge pH could be manually monitored on a periodic basis and the lime dose adjusted as required. Alternatively, if sludge were being hauled away regularly by tank truck, lime could be injected into the sludge as it was pumped into the truck. This technique was tried during the course of this program and was found to work quite well. The septic tank truck used to haul and spread lime treated sludge on the outdoor plots used a vacuum system for sludge loading. A vacuum was taken on the truck tank and sludge was pulled into the tank. The technique used to inject lime slurry into raw sludge as it was being loaded into the tank was quite simple. A suction line with a one-half inch ball valve for slurry metering was attached to an existing threaded opening in the tank sludge loading port. Then, when suction was applied, both sludge and lime slurry were pulled into the tank. Mixing occurred at the point of slurry injection and during transport to the outdoor plots. Composite samples were taken during sludge spreading operations and were found to be at pH of 12.2. This technique of lime addition could easily be applied for lime stabilization during emergency operations.

SUMMARY AND CONSLUSIONS

A process for producing lime stabilized sludge was developed and operated successfully at pilot scale. Significant reductions in pathogen populations and obnoxious odors resulted from lime treatment. Growth studies, both in a greenhouse and on outdoor plots, indicate that disposal of lime stabilized sludge on cropland would have no detrimental effects. On the basis of laboratory, pilot plant, and crop growth studies, the following major conclusions were drawn.

Laboratory Studies

- Lime dose required to raise the pH of a given sludge to a specified level was significantly influenced by the chemical characteristics of the sludge and by the technique used to mix the lime and sludge. The required amount of lime to elevate pH of a mixed primary and trickling filter sludge to 12.4 was found to vary from 4 to 10 g/ℓ as the sludges total solid concentration varied from 1.0 to 4.4 percent.
- The chemical demand for lime exerted by the chemical components of the sludge caused a pH decay over time; although an oversupply of OH ions by addition of excess $Ca(OH)_2$ can retard this decay.
- Significant reductions in indicator and pathogenic bacteria were achieved by lime treatment of sludge to $pH \geq 12.0$.
- Lime treatment had a deodorizing effect on sludge. The threshold odor numbers in a sludge with a 2.0 percent total solid concentration was reduced by 88 percent at $pH \geq 11.2$. An 83 percent threshold odor number reduction was attained in a lime treated sludge with a 4.4 percent total solid concentration at $pH \geq 11.6$.

Pilot Plant Studies

- The lime dose required to achieve $pH \geq 12.0$ is significantly affected by the chemical demand exerted by the chemical components in the sludge liquid and solid phases, and the long term chemical demand is a function of the sludge total solids concentration.
- Continuous processing of the sludge to $pH \geq 12.0$ reduced the pathogenic bacteria and indicator organism population by ≥ 99 percent.
- Lime treatment significantly improved the sludge's settling characteristics.

Growth Studies

- In a silt loam type soil, application of sludge appeared to increase permeability with water; whereas, sludge application to a sandy soil appeared to decrease permeability.
- Application of lime treated sludge did not significantly increase soil pH. The pH level of the soil-sludge mixtures was lower after plant growth than before.
- Application of lime treated sludge to cropland did increase the concentration of nutrients available to plants.
- Application of the proper amount of lime treated sludge appeared to improve soil productivity as indicated by mass of plant material produced.
- Excessive concentration of nutrients by plants did not appear to be a problem. The concentration of iron was consistently higher in the soils which received sludge applications.

ACKNOWLEDGMENTS

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THERMAL DEGRADATION OF SLUDGES*

by

Robert A. Olexsey**

INTRODUCTION

To a sanitary engineer, a simplified wastewater treatment plant resembles the process outlined in Figure 1. Hydraulically, it is a steady state, steady flow process. "X" gallons of relatively pure water come in and "X" gallons of relatively more pure water go out.*** Along the way, by-products in the form of semi-solid contaminants are discharged.

The specialist in ultimate disposal is primarily concerned with the handling and disposal of these semi-solid materials collectively known as sludge. To an "ultimate disposalist" the same wastewater treatment plant could be schematically depicted as in Figure 2. This analogy to a sludge factory is appropriate. Sludge is a material manufactured at the treatment plant. The raw materials are sewage and any chemicals that may be added at the plant. The manufacturing processes are the various treatment steps such as sedimentation, aeration, and advanced waste treatment. At the sludge factory, the relationship between the sludge product and the effluent by-product is not an obscure one. The larger the quantity of product manufactured, the higher the quality of the by-product discharged.

From a manufacturing engineering standpoint, we are concerned with producing as much sludge as possible as quickly and efficiently as possible and

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***English to Metric unit conversions in Table 8 at end of text.

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then disposing of this sludge as rapidly and economically as possible. From a manufacturing engineering standpoint it would seem a paradox to produce a product whose fate is to be swiftly destroyed. However, such is the nature of the material produced at the sludge factory that even in today's booming commodity market, it can safely be stated that the world's poorest man would be he who has cornered the sludge market. Although some of sludge's relatives in the waste family, most notably solid waste, have been attracting increased attention as potential sources of energy and materials, sewage sludge remains a material of negative economic value.

So alien is this concentration on sludge production and disposal that as recently as 1967 the WPCF Manual of Practice for Sewage Treatment Plant Design devoted a total of 30 pages out of 375 to the topic of sludge disposal (1). Lately, however, we have come to realize that sludge handling absorbs 35 percent of the capital costs and 55 percent of the annual operation and maintenance costs of a wastewater treatment plant (2). These cost shares would increase significantly if disposal were universally carried out in a proper manner.

Since the only certainty our disposalist is faced with is a larger quantity of sludge to deal with tomorrow than today, he will naturally seek out methods that will promptly put today's sludge out of sight and out of mind. Although land disposal of sewage sludge remains the most common method of application, there are increasing pressures on the disposalist to employ on-site reduction and disposal methods. The cost of land is certainly one of these factors. Health concern is obviously another. However, the unquantifiable desire simply to get rid of the stuff is probably the main driving force that pushes our disposalist to seek alternatives, often more expensive, to land disposal. Most often these alternatives take the form of some type of thermal processing or thermal degradation technique.

INCINERATION

Below is a list of some properties that are typical of most sewage sludges. The most notable parameter is that raw sewage sludge, as dirty as it may appear, is still essentially water. Also, this sludge, particularly the secondary component, has such an affinity for its water, that it clings tenaciously to it.

1. Greater than 95 percent water
2. Fierce inclination to retain water
3. Difficult to handle
4. Odorous at operational temperatures
5. 30 percent or greater nonvolatile materials in solids
6. Corrosive to incinerator construction materials and air pollution control equipment

After carefully assessing this information and having performed a rigorous engineering analysis of the merits of the process we decide to dispose of the sludge in a most practical manner: we burn it. At least we have been burning it. Since 1935, when the first sewage sludge incinerator was installed, incineration has grown in usage so dramatically that it now accounts for over 25 percent of the tonnage of sludge produced in the United States (4,5). In 1974, this translates into approximately 1.3 million tons of sludge incinerated and by 1985 the current trend points toward a figure of over 4.0 million tons incinerated (5). Table 1 highlights the projected growth of incineration as a sludge disposal medium. Note that a large contribution to the percentage increase in sludge incineration can be expected to come from the densely populated coastal areas that have been extensively utilizing ocean disposal. In fact, sludge incineration has become such a way of life that it has become almost axiomatic in consulting engineering circles to recommend sludge incineration for treatment plants serving over 15,000 people (6).

Since incineration will continue to gain prominence as a sludge disposal method, let us discuss it. Theoretically, sludge incineration can be defined as: "A thermal degradation process through which the organic fraction is converted to carbon dioxide and water vapor and the inorganic fraction to an inoffensive ash in an environmentally acceptable manner for the purpose of volume reduction and solids sterilization (7). Unfortunately, and all too often, a practical definition of sludge incineration could be: "A basically uncontrolled process for converting a single medium pollution problem into a three media pollution problem with the use of a minimum of thermal efficiency at a maximum of cost." Although incineration facilities are seldom designed in this manner they are often operated under less than optimal design conditions.

Assuming our disposalist is undaunted by the negative aspects of sludge incineration and he proceeds on his course, he very quickly encounters a fact of life that firemen have handed down from generation to generation from antiquity: water doesn't burn. Remembering that raw sludge is received in a mixture that is 95 percent water, in order to burn the 1.3 million tons of municipal sewage sludge that will be burned this year, 25 million tons, or 50,000,000,000 (50 billion) pounds, or 6 billion gallons of retained water must be dealt with. It's no wonder that people will go to such exotic means to settle out, pump out, filter out, press out, flush out, dry out, squeeze out, boil out and even freeze out the water. In fact, such is the intensity of the dewatering campaign that plants will spend from $5 to $35 for every dry ton just to remove about 80 percent of the water (8,9).

Despite its operational shortcomings, environmental side effects, and high cost, incineration has its attractions. It alters the problem by changing the offensive mass into a less offensive mass of smaller volume. It will provide an immediate resolution of the problem tomorrow as well as today. Although it is not an ultimate disposal technique, it does prolong the usefulness of the land areas as a disposal site. Although the incinerator is a depreciable asset, it is not a depletable resource. When the land area has accommodated its capacity it is finished as a disposal site. When the incinerator has accommodated its hour's capacity, it is ready for the next hour's problem. Finally, many communities simply have no other viable option. If asked why they incinerate sludge, they would respond: "Because it's there."

THE MULTIPLE HEARTH

Having decided on combustion as the most reasonable approach to our disposal problem, we must now decide what to burn our sludge in. Mostly, we burn it in a multiple hearth furnace. In 1972, there were about 125 multiple hearth sludge incinerators operational in the United States, out of a total of approximately 220 installations incinerating sludge (10,11).

A typical multiple hearth incinerator looks like the unit described in Figure 3 (12). If the design looks quaint, it's because multiple hearths have been around since 1889 when the first successful unit for roasting pyrite ore was built (13).

The furnace consists of a refractory-lined cylindrical steel shell.

Inside the shell are a number of annular hearths stacked horizontally at a fixed distance above one another. A centrally located cast iron shaft runs the full height of the furnace and supports two or four cantilevered rabble arms above each hearth. Each arm contains several rabble teeth that rake sludge spirally across the hearth, below the arms, as the arms rotate with the central shaft. The sludge, optimally in excess of 25 percent solids, is fed in at the periphery of the top hearth. It is then raked by the rabble teeth towards the center to an opening through which it falls to the next hearth. Here the sludge is rabbled outward to the periphery and so on down the furnace. The sludge and gas streams move countercurrent to one another, the sludge passing down the furnace and eventually becoming ash and the combustion air moving upward over each hearth and exiting as flue gas at the top hearth (14).

There are three operating zones in a multiple hearth: sludge is dried; it is burned; it is cooled. Gas temperatures range from 800° F on the top hearth, to 1650° F on the burning hearths, to 300° F on the bottom hearth (15).

Multiple hearth units come in sizes ranging from 3 feet in diameter and 4 hearths to over 22 feet in diameter with 12 hearths (7,16). Hearth areas range from 85 square feet to 3100 square feet. At 25 percent solids, a rule of thumb for hearth loading is 8 pounds of wet sludge per hour per square foot of hearth area (16).

Volume reduction achievement in the multiple hearth incinerator is typically in the 90 percent range (10,12,17). The following list describes the type of chemical content of ash that can be expected (17).

Content	Percent of Total
Silica (SiO_2)	25-30
Alumina (Al_2O_3)	10-13
Iron Oxide (Fe_2O_3)	9-10
Magnesium Oxide (MgO)	2-2.8
Total Calcium (CaO)	30-37
Available CaO	1-2
Phosphorous Pentoxide (P_2O_5)	7-10
Loss on Ignition	0.5-1.0

Ash handling systems can be wet or dry, but there is an increasing tendency to use the hydraulic handling lagoon approach because of the dust problems associated with the pneumatic handling-landfill approach.

Figure 4 is a flow and equipment diagram for a multiple hearth incineration system (17). Thermal efficiency of the unit is enhanced by the recycling of the exhaust air used to cool the working parts of the furnace. However, to ensure complete thermal oxidation, excess air to the extent of 100 percent over stoichiometric is often supplied. The diagram in Figure 4 does not include an afterburner for exhaust gas deodorization. In the literature one body of opinion holds that this deodorization is not necessary (18). It is stated that odor-producing compounds are not distilled from the sludge on the top hearths, and there is much reference to the "thermal jump" phenomenon that prohibits this distillation (17,19). However, since a malfunctioning thermal jump has been a difficult thing to explain to neighbors, conservative design practice has been to incorporate an afterburner arrangement to ensure that the gases are heated to a temperature of 1400° F (12,20,21).

The best thing that can be said about a multiple hearth is that it works, which is the best thing that can be said about any machine. It is simple in operation, of proven durability, and is flexible enough to burn fluctating loadings of a wide variety of materials, including grit, screenings and skimmings.

THE FLUIDIZED BED

If we do burn sludge but do not burn it in a multiple hearth, we most likely do it in a fluid bed reactor. Fluid bed technology was originally developed for catalyst recovery in oil refining operations (20). Fluid bed incineration was first applied to sewage sludge disposal in 1962 (22). By 1972 there were over 25 fluid bed sludge disposal installations in the U. S.(10). Figure 5 is a cross section of a vertical fluid bed unit (20). This is the most common type, although "flatter" horizontal types are also used.

The fluid bed unit utilizes a bed of inert materials capable of completely oxidizing all the organic material in the sludge. The technique is based on the principle that when solids are suspended in an upward moving stream of gases, the mixture possesses the characteristics of a liquid. The properties of this fluidized bed in terms of mixing and heat transfer are utilized to effect both rapid mixing and almost instantaneous drying and incineration of the organic material in the sludge (23).

For combustion of sludges, suspension of the relatively fine particles, normally less than 1/4 of an inch, of the solid material is effected in a stream of rising gas in the cylindrical reaction vessel. The velocity of the gas flow is regulated so that a vigorous mixing action is maintained in the fluidized bed. Bed temperature is normally maintained at 1300° to

1500° F to insure complete oxidation and destruction of odors (24).

Figure 6 is a schematic for a fluid bed system (23). The sludge can be either injected into the bed area as shown in the diagram or pneumatically sprayed in a controlled dispersion pattern into the "freeboard" zone above the bed material. The material that constitutes the bed material is generally silica sand or some other inert material inserted into the reactor at initial startup and replenished periodically. However, the bed could be made up of inorganic materials from the sludge feed, so long as these materials do not melt at operating temperatures and prevent fluidization of the bed. By varying the velocity of the air stream, the operation of the furnace can be adjusted so that these inorganic materials are either entrained in the exhaust gases or retained in the bed itself (24).

Although the bed has to be preheated to 1250° F before the sludge can be introduced at startup, the bed does serve as an excellent heat capacitor (12). In one series of tests it was found that a bed that had been shut down at 1400° F lost only 8° F per hour (25). The bed itself has a heat capacity of 16000 BTU/cubic foot, which is another way of saying that the sludge dries and burns pretty quickly (26). Due to the high exit gas temperatures of 1400° F, the thermal efficiency of a fluidized bed is generally lower than that of a multiple hearth unit. One study found the thermal efficiency of the fluid bed furnace to be 40 percent as opposed to 55 percent for the multiple hearth unit without an afterburner (27). Thermal efficiency can be improved through the addition of an inlet air preheater that operates off waste heat from the exhaust gases and preheats the incoming air to 1000° F (28). However, since these preheaters can represent as much as 15 percent of the capital investment of the plant, and since maintenance costs for the preheating systems have been formidable, these optional subsystems have most often been deleted from fluid bed installations in the cheap fuel era (20,29). Operating pressure in the bed area approximates 2 psi and fluid bed units have demonstrated the capability to operate at excess air rates of as low as 20 percent over stoichiometric (12,30).

Fluid bed furnaces for sludge combustion are commercially available in freeboard diameter (O.D.) sizes ranging from 4 to 16 feet with functional cross sectional areas of from 3 to 150 square feet (11). Sludge handling capacities can vary from about 200 to 2000 lb. of dry solids per hour at 25 percent solids and operational units have accommodated as much as 3400 dry pounds per hour at feeds averaging 50 percent solids (23). However, fluid bed utilization has been confined primarily to smaller installations, probably because the fluid bed represents a relatively new entrant in the sludge incineration equipment field (10).

The most prominent advantages of the fluidized bed combustor are simplicity of operation, since the reactor has no moving mechanical parts, and odor-free operation. Also, the violent motion of the bed and high temperature operation prevent the formation of clinkers in the form of metallic salts that are an operational nuisance to many multiple hearth facilities (20). An historic disadvantage has been the difficulties experienced in achieving a proper solids fluidizing distribution in the larger sized units (31).

OTHER MODES OF INCINERATION

Of less importance, because they are less widely applied are the flash dryer-incinerator, the rotary kiln, and the cyclone furnace.

Figure 7 is a flow diagram for a flash drying and incineration system (32). There are currently about 50 of these installations in the United States, but their numbers are declining (10). The process was first applied to the drying of sewage sludge in 1932 and was very popular in the 1950's (11,33).

As is evidenced from Figure 7, the process is more complex than conventional incineration. There are really two subsystems: one for drying and one for burning the sludge. In the drying cycle, the wet sludge is mixed with previously dried sludge to reduce the moisture content of the whole mixture to about 35 percent (21). The mixture then meets hot gases at approximately 1100° F from the incinerator (or burner) and is then fed to a cage mill (12, 21). By the time the sludge leaves the cage mill it has been dried to a moisture content approximating 10 percent in a matter of seconds (20). The drying gases then carry the sludge particles upward to the cyclone where centrifugal action separates them. The dried sludge is then discharged through an airlock with a portion going back to the mixer while the remainder is either burned or used as a fertilizer (21).

It is this flexibility in end use that accounted for the flash dryer's popularity. However, in recent years, the market for dried sludge as a fertilizer has been depressed to the point where operational costs cannot be recovered. If the fertilizer cannot be sold, then the advantage of flexibility has disappeared and what is left is an expensive incineration system. In fact, since most installations were constructed for drying without incineration, operational costs have been pushed upward because pure fuel has been used to dry the sludge (10,11). Even with limited heat recovery potential, the proposition is a costly endeavor.

Less widely used is the rotary kiln, depicted schematically in Figure

8 (34). It operates similarly to the multiple hearth in that sludge is dried in the upper area and burned in the lower region. Like the flash drier, it has been used most widely for sludge drying but has been also applied to combustion of sludge with refuse. Units are available in sizes ranging up to 2400 lb/hr capacities (35). A rotary kiln unit has demonstrated the flexibility to dispose concurrently of liquid tars, solid wastes, and chemical wastes (36).

The newest type of commerically available incinerator is the cyclonic reactor, represented pictorially in Figure 9 (37). The principle of the cyclone furnace is to use a vertical cylindrical combustion chamber designed so that the air velocity and the physical locations of the air inlets impart a rapid spinning motion to the column of gas inside the chamber. Due to this spinning motion, centrifugal force effects a separation of cold dense air from the hot, light combustion products. A central vortex is created that is so hot that all organic matter is destroyed before the gas stream leaves the chamber. Particulate matter is thrown outward to the walls and retained inside the furnace by centrifugal force.

The best application for the cyclonic reactor is for smaller treatment plants because of their simplicity of operation. The largest unit commercially available is 25 feet in diameter with a capacity of 7 tons per hour (38). Operation is in the odor-free temperature range and the relatively clean nature of the exhaust gas is particularly conducive to the use of a heat exchanger to recover waste heat (39). Cyclone furnaces have accommodated sludges with solids contents as low as 13 percent with considerable addition of auxiliary fuel (38).

WHAT GOES IN

Figure 10 describes a sludge processing system and what goes in and comes out of it. We will discuss these inputs and outputs for a sludge incinerator.

Three commodities go into a sludge disposal system: money, sludge, and fuel. These three are interrelated, but we will discuss each separately.

COSTS

Mostly, what goes into a sludge disposal system is money. Costs of incineration are of three types: preparation, capital, and operational (40). Preparation costs are those incurred in preparing the sludge for incineration and include capital and operational costs for thickening, transporting, dewatering, and storing. Capital costs are construction costs amortized over 20 years at 7 percent. These are fixed costs and must be paid whether we operate or not. Operational costs include labor, maintenance, fuel, and equipment requirements for operating the incinerators and are equivalent to variable costs incurred in a production facility.

First, let us put incineration costs into perspective with other disposal methods. Incineration is expensive, as is shown below (41):

Method	Total Costs ($/Dry Ton)*
1. Disposal as liquid soil conditioner	20
2. Dewatered sludge as soil conditioner	32
3. Heat drying	64
4. Lagooning	15
5. Landfilling dewatered sludge	32
6. Barging to sea	15

7. Pipeline to sea	14
8. Incineration	32

* Cost figures do not include any credit for sale of products or by-products.

Cost figures include preparation costs such as dewatering and digestion but do not include costs for land which may vary widely. No offsetting compensation is made in the cost figures for revenue for the sale of products such as soil conditioner from heat drying operations because any such credits could not be universally applied. In addition, no costs are included for disposing of incinerator ash since no firm cost data are available.

For comparison purposes we will assume that, regardless of the combustion process chosen, sludge preparation costs will be identical. This is not completely true since the newer fluid bed installations tend to use centrifuges for dewatering whereas the older multiple hearth complexes rely primarily on vacuum filters (11). However, average dewatering costs will be close to $12 per ton and handling and storing processes will be similar.

From Table 2 for a multiple hearth plant, without exhaust gas deodorization, capital costs can range from $22 per dry ton for a 10 TPD plant to about $8 per dry ton for a 100 TPD plant. Operational costs of from $8 to $13 per ton give total costs for multiple hearth incineration of from $16 to $35 per dry ton exclusive of deodorization and dewatering. When the $4 to $10 per ton costs for deodorization are added, the costs accelerate to $20 to $45 per ton.

Cost data for fluid bed units are more elusive, but there are references to total costs of from $33 to $52 per dry ton for small plants (43). Data for larger plants are inconclusive, but indications are that costs are competitive with the multiple hearth installations (23). A key factor is the installation of the inlet air preheater, which will increase capital costs but decrease variable operating costs because of decreased fuel consumption. This lever relationship would tend to indicate that the preheater may be a good investment at the larger plants and a function of fuel costs and availability at the smaller plants.

Where sale of the dried product is not possible, heat drying is simply not economical. It has been estimated that incinerating sludge in a system designed for drying and/or incineration would require an increase in basic unit costs of 17 percent and that, under the least favorable market conditions, from 25 to 40 percent of the annual sludge production must be sold to justify selection of an incineration-drying cycle over straight incineration (44).

SLUDGE

Presently, about 75 percent of the U. S. population is serviced by sewage treatment plants. Sixty-five percent of the sewage from these plants is treated to the secondary level and only 2 percent is treated to the tertiary level (45). Current Federal standards call for total secondary treatment by 1977 and escalate to no pollutant discharge by 1985 (46).

Even partial implementation of the Federal standards will greatly affect sludge disposal technology if only by influencing the sheer quantities of materials to be handled. Secondary treatment doubles the amount of sludge generated by primary treatment alone (47). Advanced waste treatment produces 25 percent more sludge than secondary treatment.

In addition, these greater quantities of sludges will be harder to work with. Whereas primary sludge is generally received with a moisture content of 95 percent, activated sludge is frequently 98 percent or more moisture (48). Volatile content, or fuel value, of the sludge can also be expected to suffer as more inert chemicals appear in the sludge mixtures with the use of advanced treatment processes.

FUEL

In 1974, fuel is the topic of the year and fuel is an increasingly important aspect of sludge combustion. In fact, the new prominence of fuel as a design and operational parameter will probably cause many people to take a hard look at conventional sludge incineration.

Sewage sludge itself is autocombustible at approximately 30 percent solids content for a mixture of equal parts of primary and secondary sludge (20). Dry raw volatile sewage solids have an average heating value approximating 10,000 BTU/lb (7). Since volatile solids can range from 60 to 80 percent of total solids by weight, a "typical" sludge has a heating value of from 6,000 to 8,000 BTU/lb. of dry solids. Values for various sludges are presented in Table 3 (7). Digestion and the presence of grit, ground garbage, and grease can alter the heating value of sludges.

Sludges cannot always be dewatered to achieve a solids content in excess of 30 percent. In fact, obtaining a 20 percent solids cake with secondary sludge is often a formidable task. Under these circumstances, additional heat from auxiliary fuel must be added to the furnace to evaporate the excess moisture in the sludge. This auxiliary fuel is normally No. 2 oil or natural gas.

Fuel requirements for combusting a 25 percent solids sludge in a multiple hearth furnace without exhaust gas deodorization can be calculated as 28 gallons of No. 2 fuel oil per dry ton of sludge (49). In actual practice, because of variations in solids contents, types of sludges, and operational efficiencies, the national average can be estimated at roughly 50 gallons per dry ton of sludge (50). With the increased use of secondary and tertiary treatment in the endeavor to attain projected discharge standards, average fuel consumption may very well double to 100 gallons per dry ton by 1985, if the present trend continues.

When fuel oil was plentiful at 12 cents per gallon, the economic burden imposed by fuel consumption was merely painful. In fact, rather than upgrade dewatering practices or invest in such items as heat exchangers to utilize waste heat, most plants simply chose to use more and more fuel in place of more efficient operation. With oil at 25 cents per gallon and rising, the economic pains become acute. With oil and gas perhaps unavailable, operation simply becomes impossible.

WHAT GOES OUT

The incinerator's two main products are ash and emissions in the form of exhaust gases. The ash portion was discussed earlier. Ash constitutes about 10 percent of the furnace feed and contains less than 1 percent combustible matter in the form of fixed carbon (17). The ash makes an adequate landfill material.

Aside from odors, which may or may not exist, the products of sludge combustion are essentially steam, carbon dioxide, and small amounts of particulate emissions and oxides of nitrogen and sulfur. The water vapor does not represent a hazard but cooling the exhaust gases to 110° F will eliminate the steam plume if this is desirable (17).

Table 4 summarizes the results of stack gas sampling for particulates NO_x, SO_2, and visible emissions from four incineration sites (51). These data are from tests on three units with multiple hearths and one with a fluid bed.

All were equipped with impingement type wet scrubbers. The figures represent ranges of sampled emissions under a variety of operating conditions. Average values would indicate that low energy wet scrubbers are an adequate air pollution control device and that sewage sludge incinerators so equipped are not a significant source of SO_2, NO_x, and particulate emissions. This is further exemplified by the fact that the same study found that maximum particulate emissions were 2.83 lbs/hr and 4.8 lbs per day per ton incinerated. Average values were much lower.

Table 5 is an excerpt from the results of analyses for metals concentration in input sludges and the resultant ash from three incinerators (52). If the metals go into the incinerator and do not come out in the ash, they must go into the scrubber as fly ash or vapor. The study further found that mercury apparently is totally vaporized into the stack gases and that lead appears in higher concentrations in the fly ash than it does in the bottom ash. Although much further study is needed, current technology would indicate that control of lead emissions is largely a function of particulate removal efficiency and control of mercury is dependent on the ability of authorities to regulate mercury discharge into receiving sewers.

Proposed EPA new source performance standards concentrate primarily on control of particulate emissions and opacity. Process weight regulations would limit emissions to 1.3 lb/ton of dry sludge (0.65 g/kg), based on a dry charging weight of 1000 lb/hr (53). The lower efficiency impingement scrubbers are adequate to meet opacity standards of 20 percent and the process weight regulations but do not represent the best control technology (53,54). Manufacturers of the high energy venturi scrubbers will guarantee outlet concentration of less than .030 gr/dscf (54).

ALTERNATIVES

We have painted a picture of incineration as a costly alternative to other sludge disposal techniques. We have shown that incinerators can represent a substantial capital investment; that they consume about one-fifth as much fuel as they do sludge; that they can contribute to air pollution if not properly operated and that control of this air pollution can be very costly. We have also described some of the aspects of incineration that make it attractive, particularly its speed and compactness. We have outlined the manner in which the advantages of incineration have manifested a definite trend towards incineration as a sludge disposal technique. Although incineration will continue to be an important part of the overall sludge disposal spectrum, increasing cost, fuel, and environmental pressures have prompted a search for disposal options that possess the advantages of on-site disposal and minimize the disadvantages.

PYROLYSIS

Pyrolysis is the destructive distillation of organic materials under pressure and heat in the absence of oxygen. It is the process that is used to produce coke from coal and charcoal from wood. Through the pyrolysis process, the organic portions of waste are reformed into lower molecular weight compounds. These compounds can be in the form of a combustible gas, tar and oil, and a solid "char" which also has an appreciable heating value (55).

Figure 11 is a flow diagram for a simple pyrolysis system (56). The "pyrolyzing" process is carried out in an externally heated closed reactor chamber. Process temperatures can be as low as 500° C (932° F) or in excess of 900° C (1652° F) at atmospheric pressures. At the lower temperatures the reaction product is predominantly solid and at higher temperatures, gas is more prevalent (57). The volatile gases can be siphoned off and used to heat the reaction chamber. The process can be thermally self-sustaining providing the heating value of the waste is high enough.

Tar and heavy oils are collected in a tar trap and additional heavy oils and liquors are condensed out of the product stream. Traces of heavy oil mists are collected in electrostatic precipitators. Gases are scrubbed clean of acids and sulfides and then returned to the retort to fuel the pyrolyzer, collected as product, or flared (57).

Most of the research and development activity in the field of waste pyrolysis has been directed towards the processing of municipal solid waste. Although no full-scale waste pyrolysis plants are currently operating, there are plans to demonstrate several commercially developed processes for municipal waste pyrolysis in the United States. Perhaps the most ambitious of these plants is planned for Baltimore, where 1000 tons per day of shredded wastes will be processed in a rotary kiln unit as described in Figure 12 (58). The system is not a pure pyrolytic operation since some air is introduced into the combustion chamber. Gases produced in the reactor will be used to fuel a boiler that will produce 200,000 pounds of saleable steam per hour.

The 200 ton per day plant under construction in San Diego is outlined in Figure 13 (59). Product yields in San Diego will be a high viscosity fuel oil, a char, and a medium BTU gas that will be used to fuel the reactor.

A third system, being demonstrated by the manufacturer in South Charleston, West Virginia, is depicted in Figure 14 (60). This 200 ton per day plant will convert unshredded refuse into a low BTU gas and a slag in a vertical shaft furnace. Slagging is accomplished through the introduction of oxygen into the lower portion of the furnace.

Although development of pyrolytic techniques for sewage sludge has not kept pace with that for solid wastes, some research has been initiated in this area. Pyrolysis of bovine waste has been performed at 900° C (1652° F) in the laboratory unit described earlier (Figure 11). For this test the cow manure was dried to 3.6 percent moisture content, although fresh manure contains over 75 percent moisture. As presented in Table 6 the pyrolysis operation yielded a gas fraction of 38.5 percent of feed, an oil fraction of 5.8 percent, and a char residue of 36.3 percent (61).

Successful pyrolysis of sewage sludge alone will depend on the ability of the reaction to generate enough fuel to dry the incoming sludge to a moisture content approaching 5 percent. A more likely implementation will be pyrolysis of sewage sludge in conjunction with solid waste.

Since technology for pyrolysis is in a relatively infant state and since hardware and operational characteristics are not on a firm basis, cost data on pyrolysis cannot be considered reliable. One estimate places capital and operating costs of pyrolysis tentatively at about two-thirds that of pollution-free incineration (62).

Pyrolysis does have the definite advantage that air pollution control requirements are minimized. If the process can be shown to be self-sustaining, fuel consumption is not the economic burden that it is in incineration processes.

Combined Incineration with Solid Waste

A possible solution to the fuel problem may be found by combining the incineration operations of municipal solid waste and sewage sludge. Solid waste makes a good fuel, having a heating value approaching 5,000 BTU/lb as shown in Table 7 (50). Refuse is a universal commodity found in all communities at generation rates proportional to sewage sludge. It has a sulfur content well below 1 percent (63). The circumstances that tend to push communities toward incineration of sludge also work in favor of refuse incineration so that the two are often practiced in parallel. Potential economic advantages may be realized in confining capital investments to a single installation. The additional heating value and solids content provided by the solid waste may enable the combustion of a wetter sludge cake and result in lower costs for dewatering.

The major consideration in co-incineration is what to burn the material in. Good solid waste incinerators tend not to be well suited for combusting materials with high moisture content. Conversely, incinerators that adequately process sludge cannot cope with the much larger quantities and physical irregularities presented by solid waste.

Combined incineration of refuse and sludge has had limited application in the United States. The most common approach has been the utilization of the waste heat from refuse combustion to dry the sludge before it enters the incinerator. Such a system is described in Figure 15. A similar system has been employed in Waterbury, Connecticut, since 1957, where 300 tons per day of refuse and sludge are incinerated (64). Other attempts at combined incineration have been made at smaller towns in Wisconsin and Pennsylvania.

Unfortunately, the limited cost information has shown the processes to be expensive, with 1957 costs for Waterbury quoted at $26 per dry ton of sludge (21). One of the Wisconsin cities, Neenah, reported auxiliary fuel consumption of about 25 gallons of oil per dry ton of sludge. However, the co-incineration process was considered economical because it reduced the cost of hauling the material to a landfill site (65). Co-incineration has also been practiced with success in a rotary kiln unit in Louisville, Kentucky, and in a fluid bed at Franklin, Ohio.

In Europe, where fuel has always been expensive, co-incineration has been practiced for years. Although recently many plants have converted to refuse combustion for steam production, plants are still operating in England, Germany, and Sweden. Installations have ranged from multiple hearths to rotary kilns.

Although combined incineration has not been overly successful, it probably merits a closer look in the United States. It is conceivable that a well-designed and operated co-incineration plant could yield lower overall refuse and sludge disposal costs than a separate incineration program that involves heat or fuel recovery from the solid waste.

Wet Oxidation

Figure 16 depicts a thermal processing technique that alleviates the need for mechanical dewatering of sludges (20,66). Wet oxidation is a process whereby organic wastes dissolved or suspended in water are converted to water, carbon dioxide, small amounts of low molecular weight organic acids, and a very small amount of fine inorganic ash (67). Since 1912, wet oxidation has been proposed as a means for disposal of spent liquors from paper-pulp mills. However, commercial application in the paper-pulp industry did not take place until 1959. The process was first applied to sewage treatment for sludge digestion and disposal in 1960. By 1972, there had been 17 plants constructed for sewage treatment and five installed for the treatment of paper-pulp-mill and other industrial wastes (68).

The process is conducted with an excess of oxygen at temperatures ranging from 300° F to 600° F in a pressure vessel which maintains the water as a liquid. Air overpressure can range from 50 to 300 psi with resulting operating pressures from 600 to over 3,000 psi (66,67). Feed solids concentrations can be as low as one percent but optimal operation is experienced with sludges thickened to from 3 to 6 percent (20,69).

Wet oxidation has evolved into two separate processes with two distinct functions. As a disposal process, wet oxidation at temperatures and pressures at the high ends of the operating ranges can accomplish a 90 percent reduction of putrescible solids. However, maintenance of the high temperatures and pressures has made continuous operation difficult and the process has been costly.

Operation at lower temperatures and pressures is in reality a form of heat treatment. Exposing the sludge to heat and pressure coagulates the solids and breaks down the gel structure. This reduces the water affinity of the sludge solids and enables production of filter cakes of up to 50 percent solids content (66). The solid residue is sterile but a strongly oxygen-demanding liquor is formed (68). Disposal of this effluent poses a substantial problem.

Although wet air oxidation does not qualify as a completely new technique and has suffered from cost and reliability problems, it is an area of much potential. With continued development, the process could become a viable disposal technique that emits no air pollutants and is self-sufficient from a fuel standpoint.

Conclusion

For the disposalist who must deal daily with the output of the sludge factory, thermal processing of sewage sludge is a disposal option that should be approached objectively. For smaller communities sludge combustion will most probably be the most expensive disposal technique. Large cities that are faced with the prospect of transporting sludge long distances to land disposal sites may find combustion to be a more attractive venture. The aesthetic and convenience advantages of on-site disposal must be weighed against the environmental, fuel-consumptive, cost aspects of incineration. Improvements in or alternatives to the sludge incineration process may, in the near future, solve all the problems associated with current thermal processing techniques, but since each new development also produces its own new problems, life promises to remain interesting in the realm of sludge disposal.

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TABLE 1: TRENDS IN SLUDGE DISPOSAL

	QUANTITY (DRY TONS/DAY)		PERCENT OF TOTAL	
MODE	1972	1985	1972	1985
GENERATION	11,000	28,000	100	100
LAND APPLICATION	6,600	16,000	60	60
OCEAN DISPOSAL	1,650	0	15	0
INCINERATION	2,750	12,000	25	40

TABLE 2: AVERAGE U. S. 1973 COSTS FOR MULTIPLE HEARTH INCINERATION ($/DRY TON)*

	PLANT SIZE	
	10 TPD	100 TPD
CAPITAL (AMORTIZED)	22	8
OPERATIONAL	13	8
TOTAL (NO DEODORIZATION)	35	16
DEODORIZATION	10	4
TOTAL (WITH DEODORIZATION)	45	20

* DOES NOT INCLUDE DEWATERING COSTS.

TABLE 3: HEATING VALUES OF SOME SLUDGES

MATERIAL	COMBUSTIBLES (%)	BTU/LB OF COMBUSTIBLES
GREASE AND SCUM	88.5	16,750
RAW SEWAGE SOLIDS	74.0	10,285
FINE SCREENINGS	86.4	8,990
GROUND GARBAGE	84.8	8,245
DIGESTED SLUDGE	59.6	5,290
GRIT	33.2	4,000

TABLE 4: EMISSIONS FROM FOUR SEWAGE SLUDGE INCINERATORS

EMISSIONS	MINIMUM		MAXIMUM
% CO_2 - VOL% (DRY BASIS)	2.2	-	10.2
% EXCESS AIR (TEST POINT)	366.0	-	51.0
SO_2 (PPM)	1.97	-	14.26
NO_x (PPM)	11.47	-	271.63
HCl (PPM)	.621	-	11.9
PARTICULATE - FILTER (GR/SCFD)	.0127	-	.0766
PARTICULATE - TOTAL (GR/SCFD)	.0170	-	.0859
SLUDGE FEED TO FURNACE (LB DS/HR)	221.8	-	1710.0
STACK FLOW RATE (SCFM)	1170	-	10290

TABLE 5: METAL TO FIXED SOLID RATIO FROM THREE INCINERATORS (MG/G)

ELEMENT	PLANT A SLUDGE	 ASH	PLANT B SLUDGE	 ASH	PLANT C SLUDGE	 ASH
CADMIUM	0.37	0.20	n.d.	0.58	0.31	0.22
CHROMIUM	2.0	0.3	2.9	0.5	0.7	0.6
COPPER	2.6	1.3	2.5	1.7	1.6	1.6
IRON	18.0	8.9	12.0	11.0	50.0	43.0
LEAD	5.8	0.7	7.0	0.85	2.0	1.0
MERCURY	9.0*	n.d.	20.0*	n.d.	6.0*	n.d.

n.d. - NOT DETECTED

* - MICROGRAMS/GRAM

TABLE 7: COMPARATIVE HEATING VALUES OF PERTINENT FUELS

FUEL	HEATING VALUE (BTU/LB)	RATIO OF FUEL VALUES TO VALUE FOR NO.2 OIL
NO. 2 OIL	19,600	1.00
NO. 6 OIL	17,500	0.89
NATURAL GAS	22,800	1.16
BITUMINOUS COAL	13.600	0.69
WOOD (AIR DRIED)	5,500	0.28
GREASE AND SCUM	16,700	0.85
SLUDGE (DRY VOLATILES)	10,000	0.52
DIGESTED SLUDGE	5,300	0.27
DIGESTER GAS	15,400	0.79
MUNICIPAL REFUSE (20% MOISTURE)	4,900	0.25

SOURCES: PERRY'S CHEMICAL ENGINEER'S HANDBOOK, 4TH EDITION;
A STUDY OF SLUDGE HANDLING AND DISPOSAL, R. S. BURD;
CHARACTERISTICS OF MUNICIPAL REFUSE, E. KAISER.

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TABLE 8: CONVERSION FACTORS FOR UNITS USED IN THIS PAPER

FROM (ENGLISH)	TO (METRIC)	MULTIPLY BY
GALLONS	LITERS	3.785
TONS (SHORT)	KILOGRAMS	907.184
POUNDS	KILOGRAMS	0.453
FEET	METERS	0.305
SQUARE FEET	SQUARE CENTIMETERS	929.030
CUBIC FEET	CUBIC METERS	0.028
SQUARE INCHES	SQUARE CENTIMETERS	6.452
BTU'S	CALORIES	252.000
GRAINS	GRAMS	0.065

FOR TEMPERATURES: TO CONVERT FROM FAHRENHEIT (°F) TO CELSIUS (°C):

$$°C = (°F - 32) \times 5/9$$

FIGURE 1. WASTEWATER TREATMENT PLANT

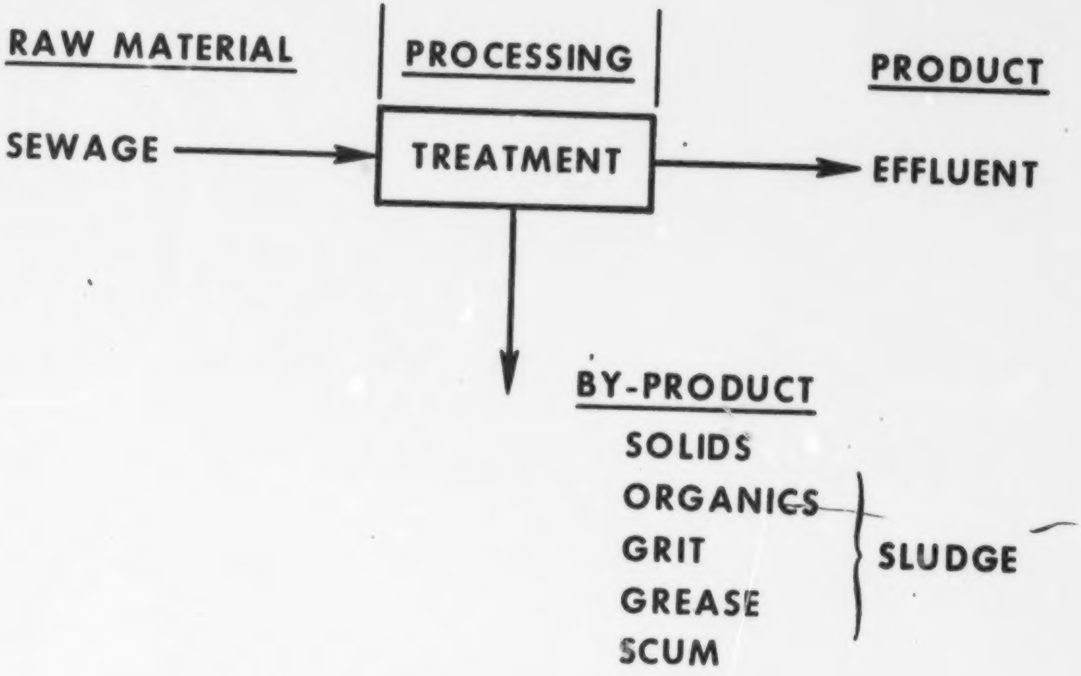
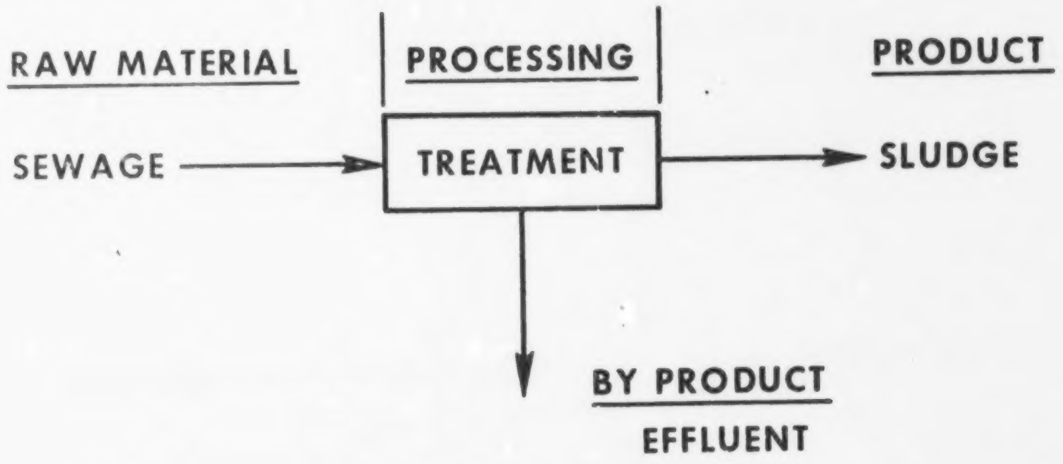

RAW MATERIAL
PROCESSING
PRODUCT
SEWAGE
TREATMENT
EFFLUENT
BY-PRODUCT
SOLIDS
ORGANICS
GRIT
GREASE
SCUM
SLUDGE

FIGURE 2. SEWAGE SLUDGE FACTORY


RAW MATERIAL
PROCESSING
PRODUCT
SEWAGE
TREATMENT
SLUDGE
BY PRODUCT
EFFLUENT

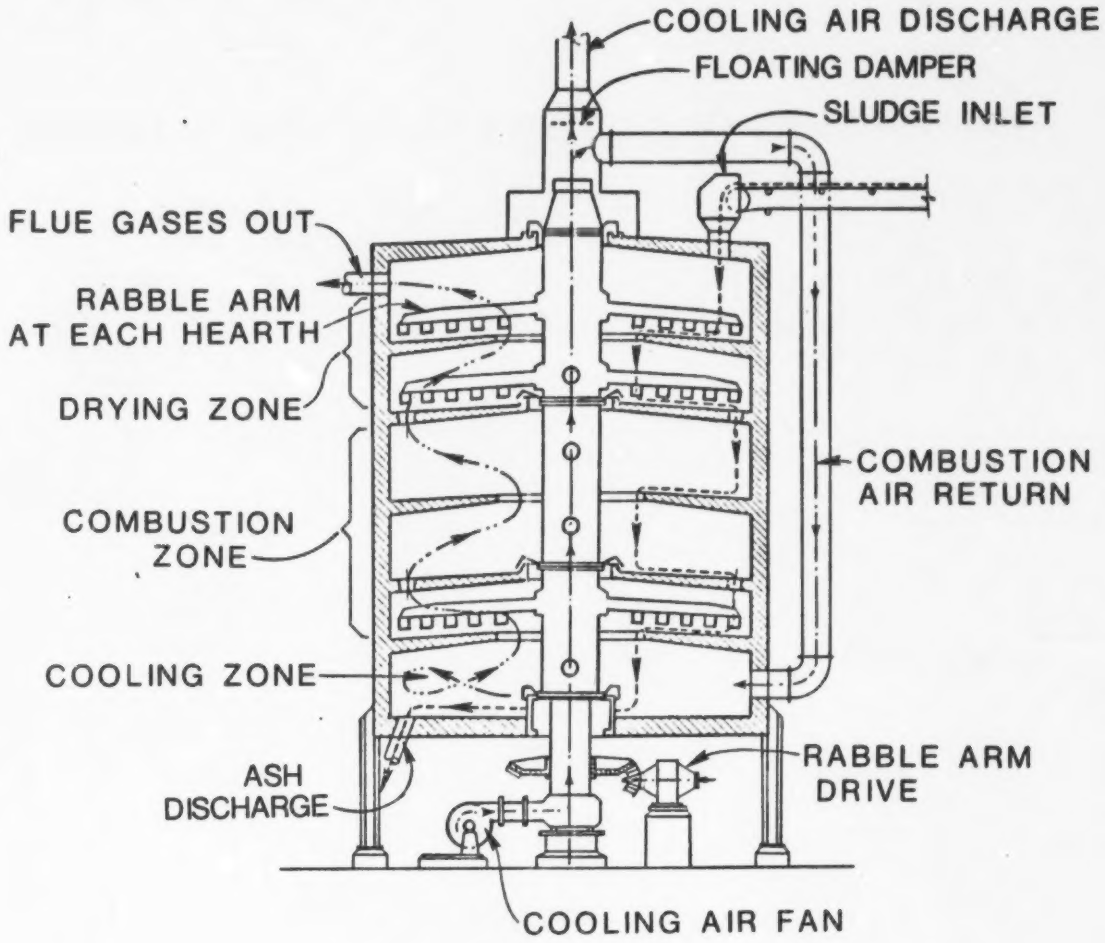

COOLING AIR DISCHARGE
FLOATING DAMPER
SLUDGE INLET
FLUE GASES OUT
RABBLE ARM AT EACH HEARTH
DRYING ZONE
COMBUSTION AIR RETURN
COMBUSTION ZONE
COOLING ZONE
ASH DISCHARGE
RABBLE ARM DRIVE
COOLING AIR FAN

FIGURE 3. TYPICAL SECTION:
MULTIPLE HEARTH INCINERATOR

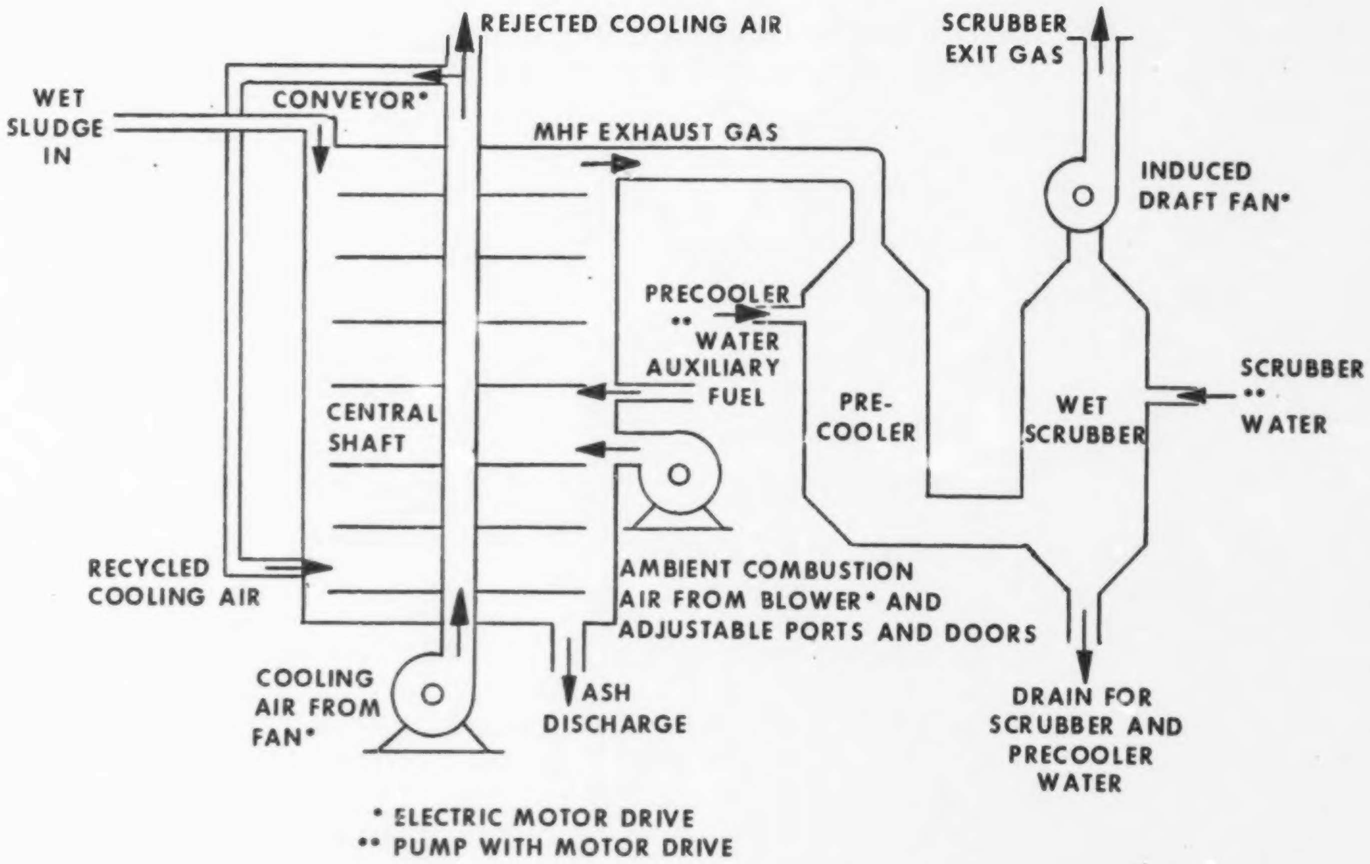

REJECTED COOLING AIR
SCRUBBER EXIT GAS
WET SLUDGE IN
CONVEYOR*
MHF EXHAUST GAS
INDUCED DRAFT FAN*
PRECOOLER ** WATER
AUXILIARY FUEL
CENTRAL SHAFT
PRE-COOLER
WET SCRUBBER
SCRUBBER ** WATER
RECYCLED COOLING AIR
AMBIENT COMBUSTION AIR FROM BLOWER* AND ADJUSTABLE PORTS AND DOORS
COOLING AIR FROM FAN*
ASH DISCHARGE
DRAIN FOR SCRUBBER AND PRECOOLER WATER
* ELECTRIC MOTOR DRIVE
** PUMP WITH MOTOR DRIVE

FIGURE 4. MHF FLOW AND EQUIPMENT DIAGRAM

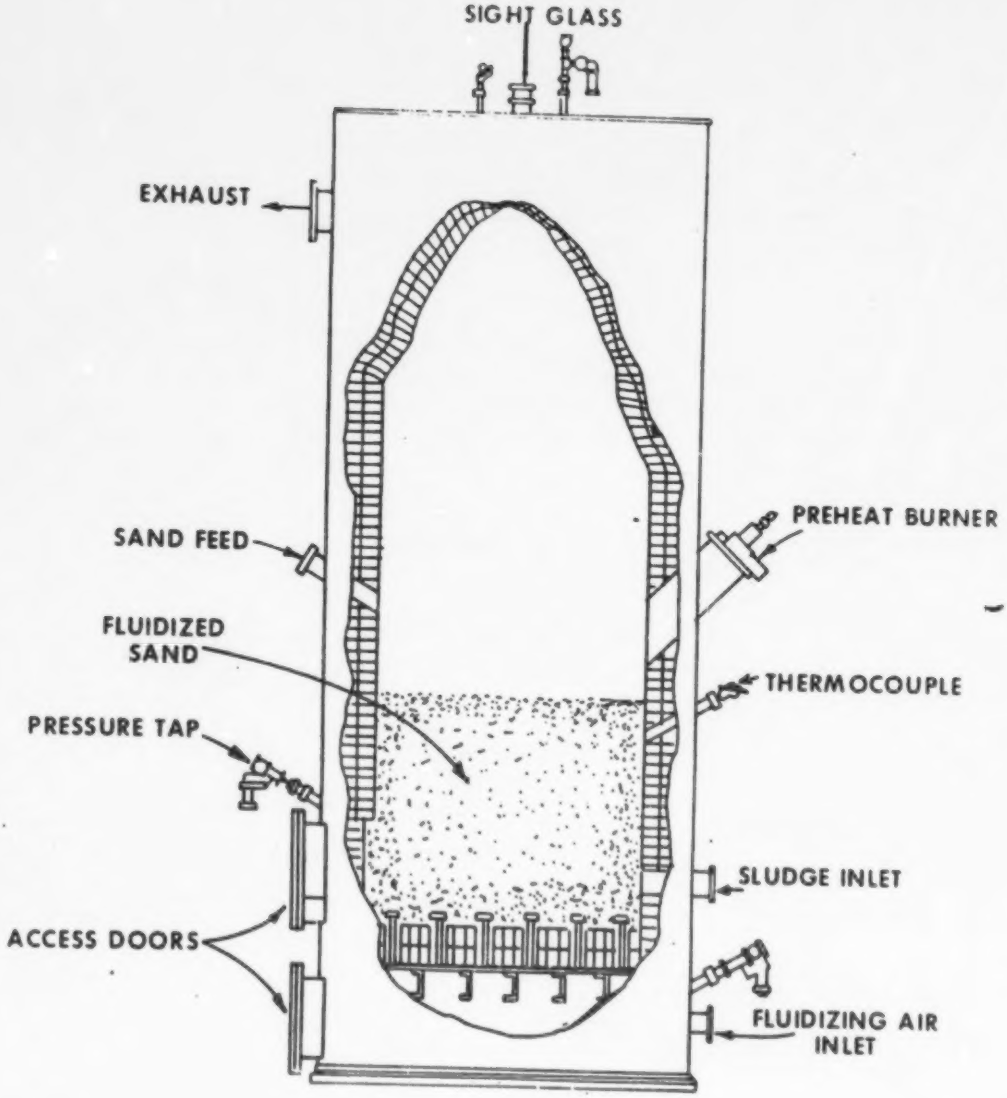

SIGHT GLASS
EXHAUST
SAND FEED
PREHEAT BURNER
FLUIDIZED SAND
THERMOCOUPLE
PRESSURE TAP
SLUDGE INLET
ACCESS DOORS
FLUIDIZING AIR INLET

FIGURE 5.

TYPICAL SECTION OF A FLUID BED REACTOR
(DORR-OLIVER, INC.)

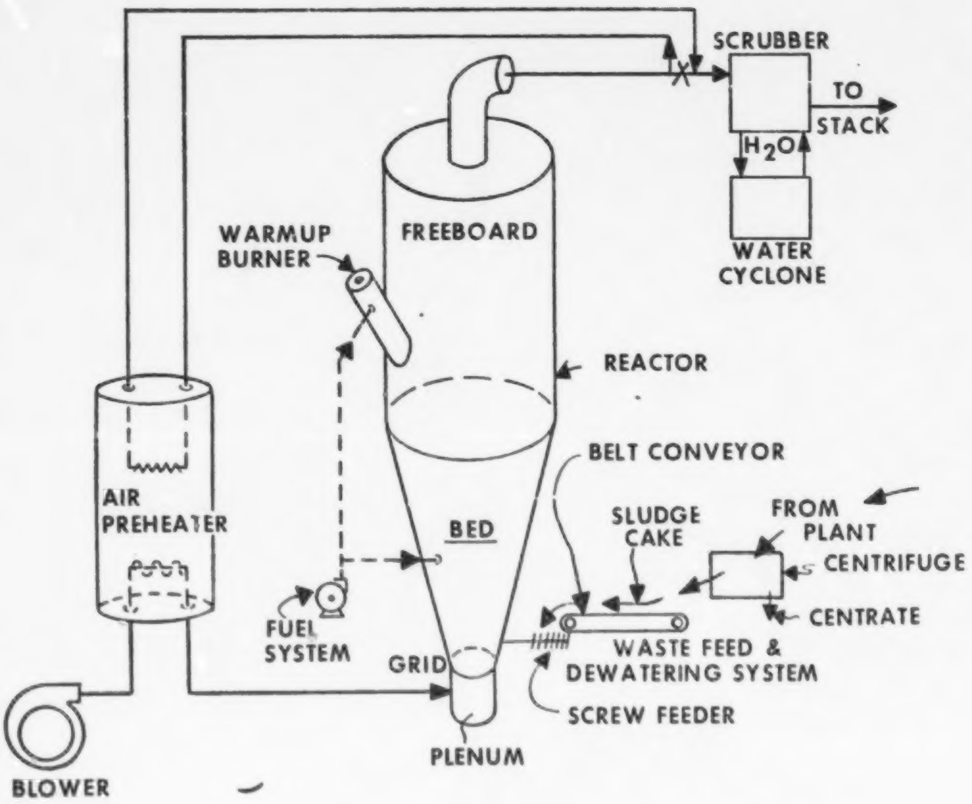
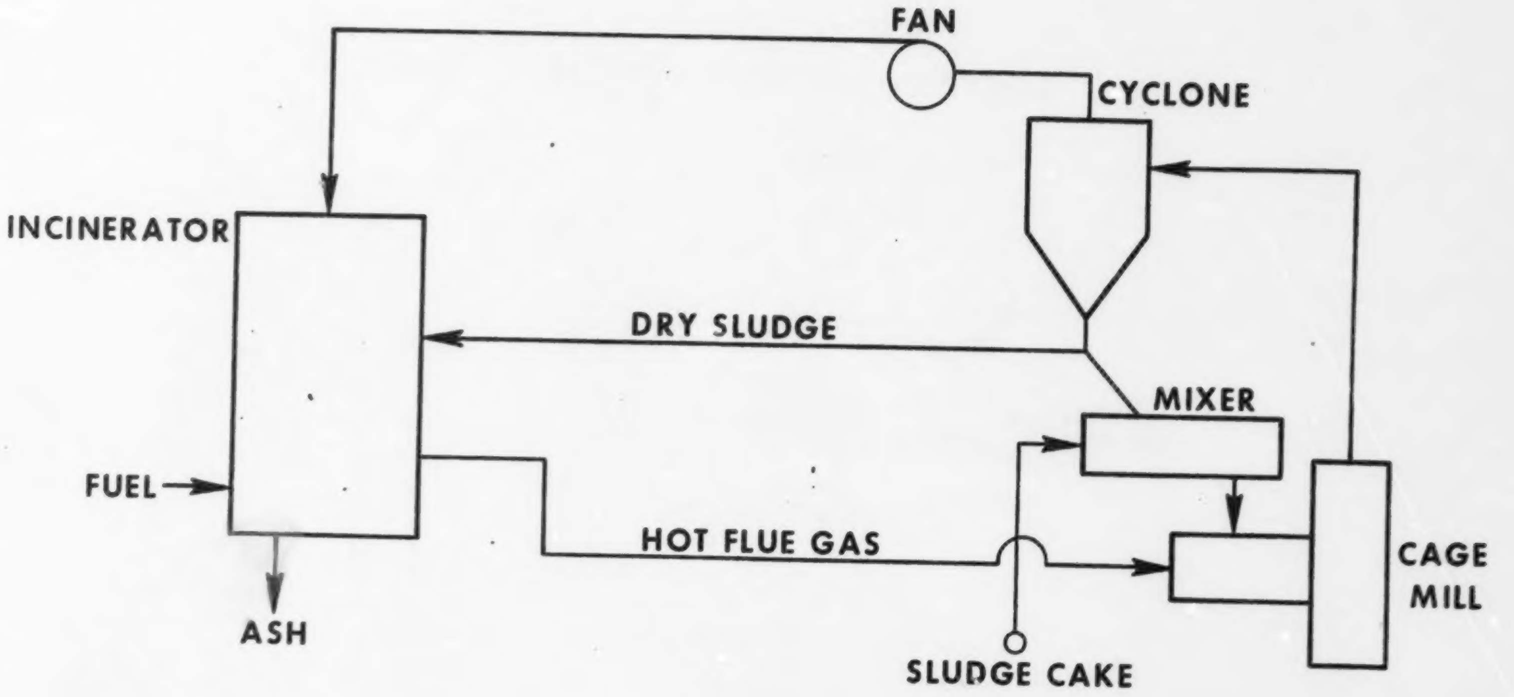

SCRUBBER
TO STACK
H_2O
WATER CYCLONE
FREEBOARD
WARMUP BURNER
REACTOR
AIR PREHEATER
BELT CONVEYOR
SLUDGE CAKE
FROM PLANT
CENTRIFUGE
CENTRATE
BED
FUEL SYSTEM
GRID
WASTE FEED & DEWATERING SYSTEM
SCREW FEEDER
PLENUM
BLOWER

FIGURE 6.

SYSTEM COMPONENTS FOR FB REACTOR

FIGURE 7. FLASH DRYER-INCINERATOR


FAN
CYCLONE
INCINERATOR
DRY SLUDGE
MIXER
FUEL
HOT FLUE GAS
CAGE
MILL
ASH
SLUDGE CAKE

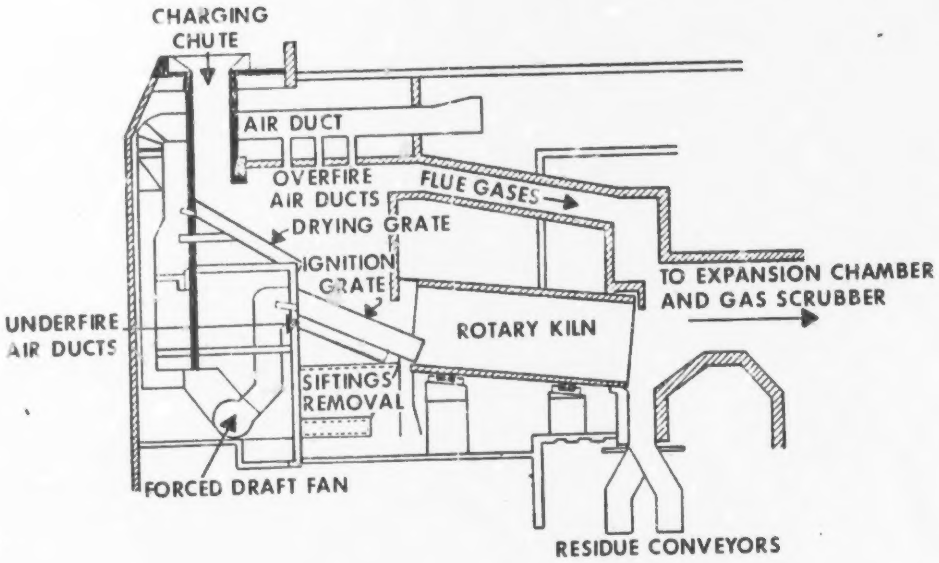

CHARGING CHUTE
AIR DUCT
OVERFIRE AIR DUCTS
FLUE GASES
DRYING GRATE
IGNITION GRATE
UNDERFIRE AIR DUCTS
ROTARY KILN
TO EXPANSION CHAMBER AND GAS SCRUBBER
SIFTINGS REMOVAL
FORCED DRAFT FAN
RESIDUE CONVEYORS

FIGURE 8. ROTARY KILN INCINERATOR

FIGURE 9. CYCLONE FURNACE

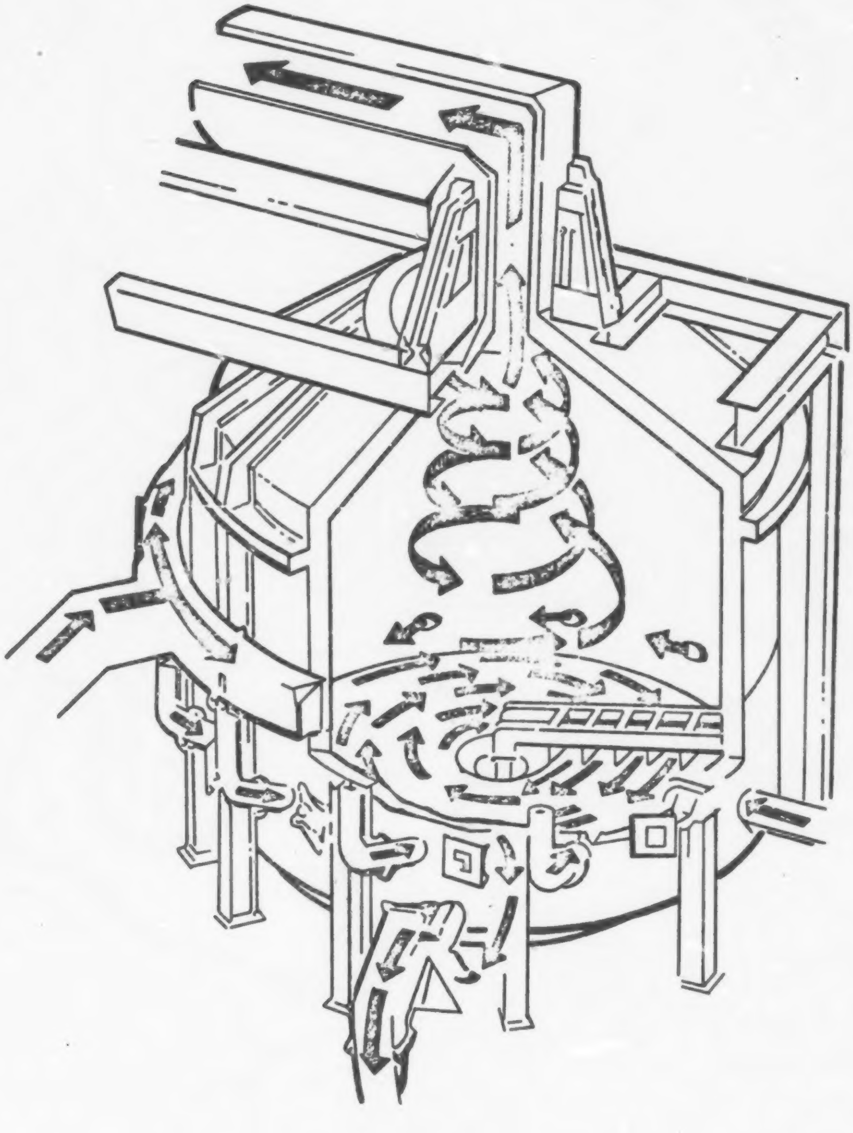


FIGURE 10. BLACK BOX CONCEPT FOR A SLUDGE DISPOSAL SYSTEM

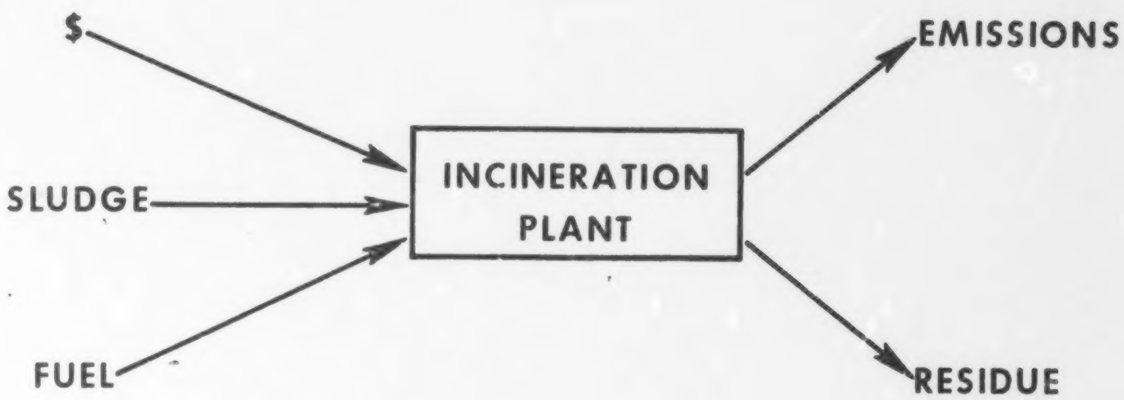
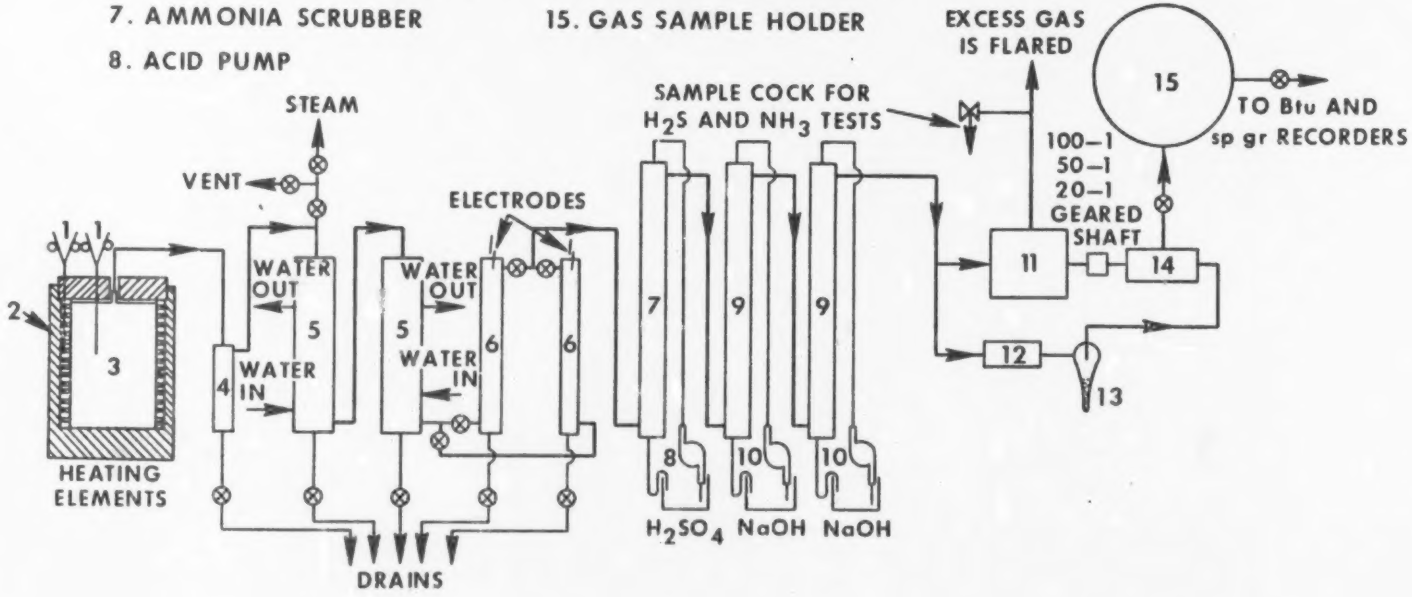

$
SLUDGE
FUEL
INCINERATION PLANT
EMISSIONS
RESIDUE

FIGURE 11. PILOT PLANT PYROLYSIS SYSTEM

LEGEND

1. THERMOCOUPLE
2. ELECTRIC FURNACE
3. RETORT
4. TAR TRAP
5. TUBULAR CONDENSER
6. ELECTROSTATIC PRECIPITATOR
7. AMMONIA SCRUBBER
8. ACID PUMP
9. CARBON DIOXIDE SCRUBBER
10. CAUSTIC PUMP
11. LARGE WET-TEST METER
12. DRYING TUBE
13. LIGHT OIL CONDENSER
14. SMALL WET-TEST METER
15. GAS SAMPLE HOLDER


EXCESS GAS IS FLARED
15
TO Btu AND sp gr RECORDERS
SAMPLE COCK FOR H_2S AND NH_3 TESTS
STEAM
VENT
ELECTRODES
100–1
50–1
20–1
GEARED SHAFT
11
14
12
13
1
1
2
3
4
5
5
6
6
7
9
9
8
10
10
WATER OUT
WATER IN
WATER OUT
WATER IN
HEATING ELEMENTS
H_2SO_4
NaOH
NaOH
DRAINS

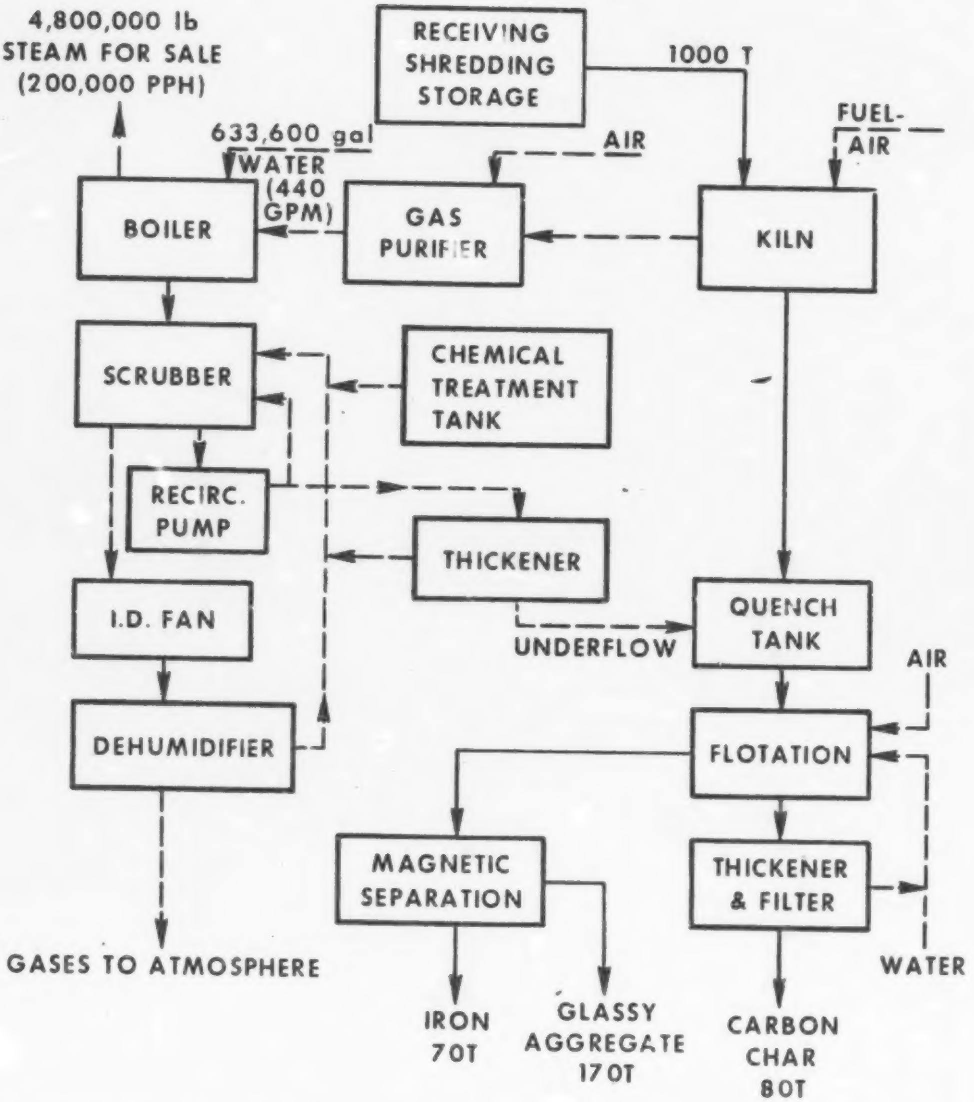

4,800,000 lb
STEAM FOR SALE
(200,000 PPH)
RECEIVING
SHREDDING
STORAGE
1000 T
FUEL-
AIR
633,600 gal
WATER
(440
GPM)
AIR
BOILER
GAS
PURIFIER
KILN
SCRUBBER
CHEMICAL
TREATMENT
TANK
RECIRC.
PUMP
THICKENER
I.D. FAN
QUENCH
TANK
UNDERFLOW
AIR
DEHUMIDIFIER
FLOTATION
MAGNETIC
SEPARATION
THICKENER
& FILTER
GASES TO ATMOSPHERE
WATER
IRON
70T
GLASSY
AGGREGATE
170T
CARBON
CHAR
80T

FIGURE 12. LANDGARD FLOW SHEET; BASIS - 1000 TONS FEED

FIGURE 13. BLOCK DIAGRAM OF THE GR&D SOLID WASTE RECYCLING PROCESS

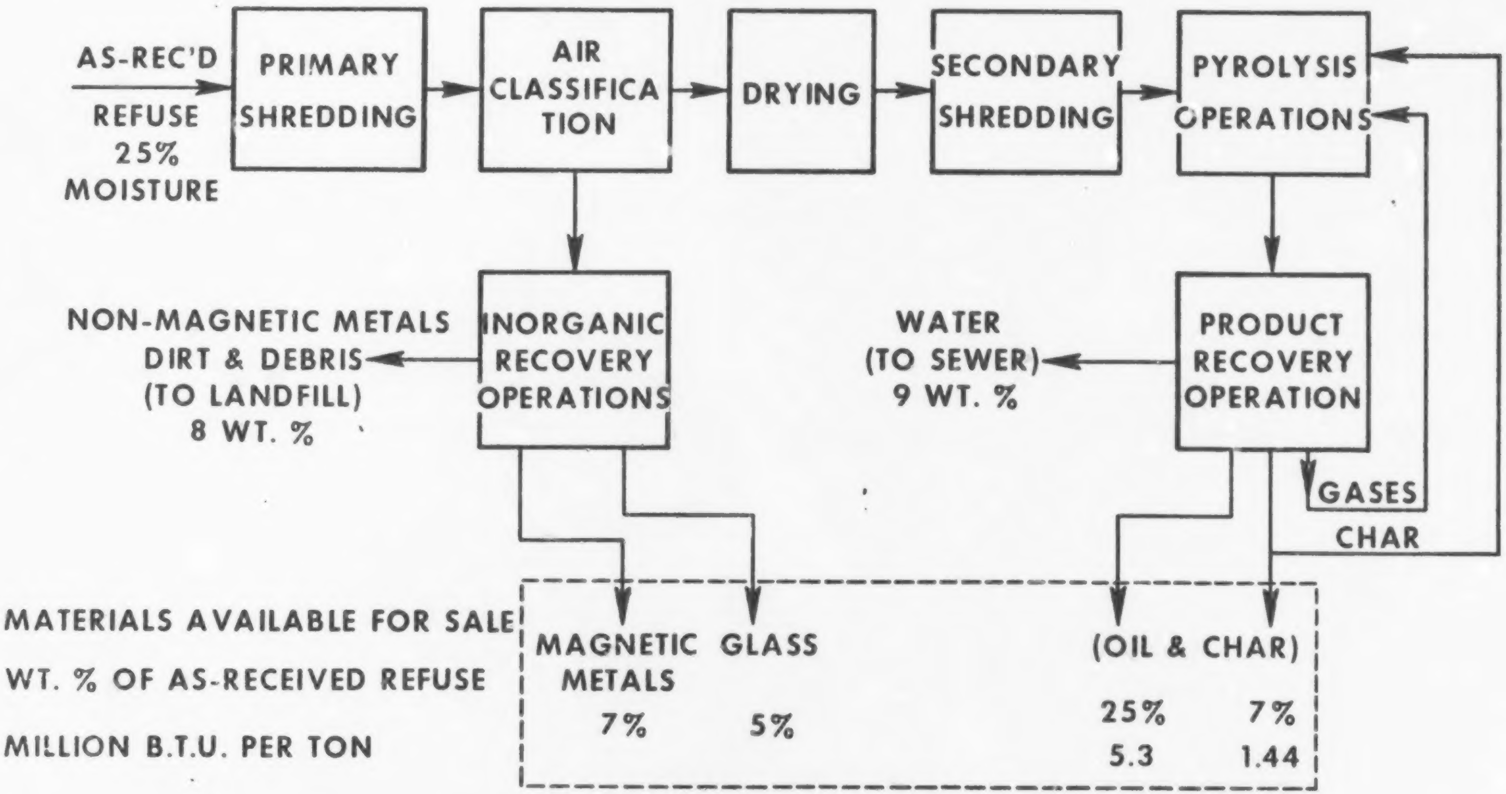

AS-REC'D REFUSE 25% MOISTURE
PRIMARY SHREDDING
AIR CLASSIFICATION
DRYING
SECONDARY SHREDDING
PYROLYSIS OPERATIONS
INORGANIC RECOVERY OPERATIONS
NON-MAGNETIC METALS DIRT & DEBRIS (TO LANDFILL) 8 WT. %
PRODUCT RECOVERY OPERATION
WATER (TO SEWER) 9 WT. %
GASES
CHAR
MATERIALS AVAILABLE FOR SALE
WT. % OF AS-RECEIVED REFUSE
MILLION B.T.U. PER TON
MAGNETIC METALS
7%
GLASS
5%
(OIL & CHAR)
25%
7%
5.3
1.44

FIGURE 14. INPUTS AND PRODUCTS OF PUROX SYSTEM

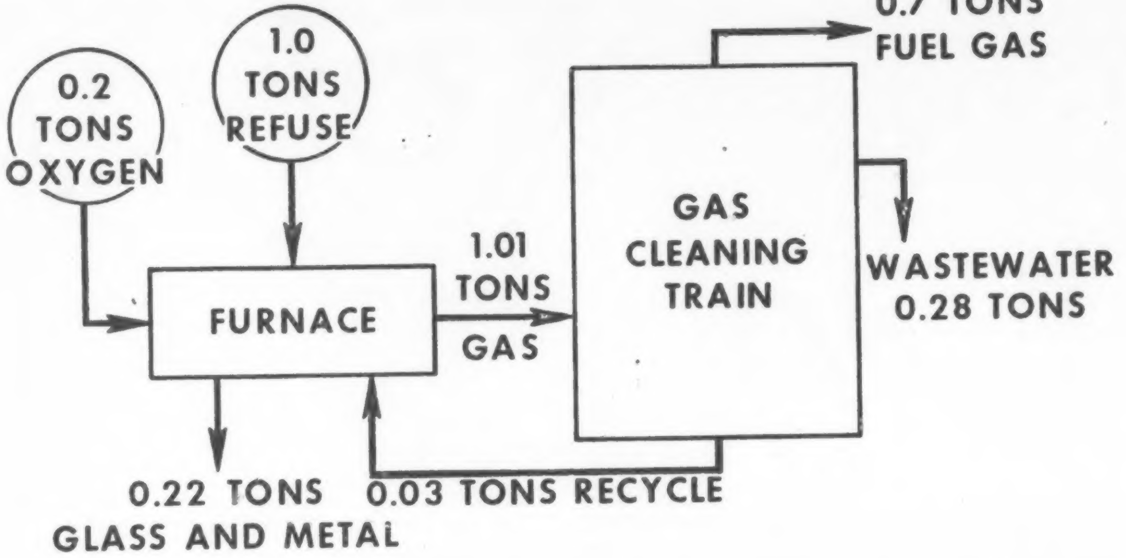
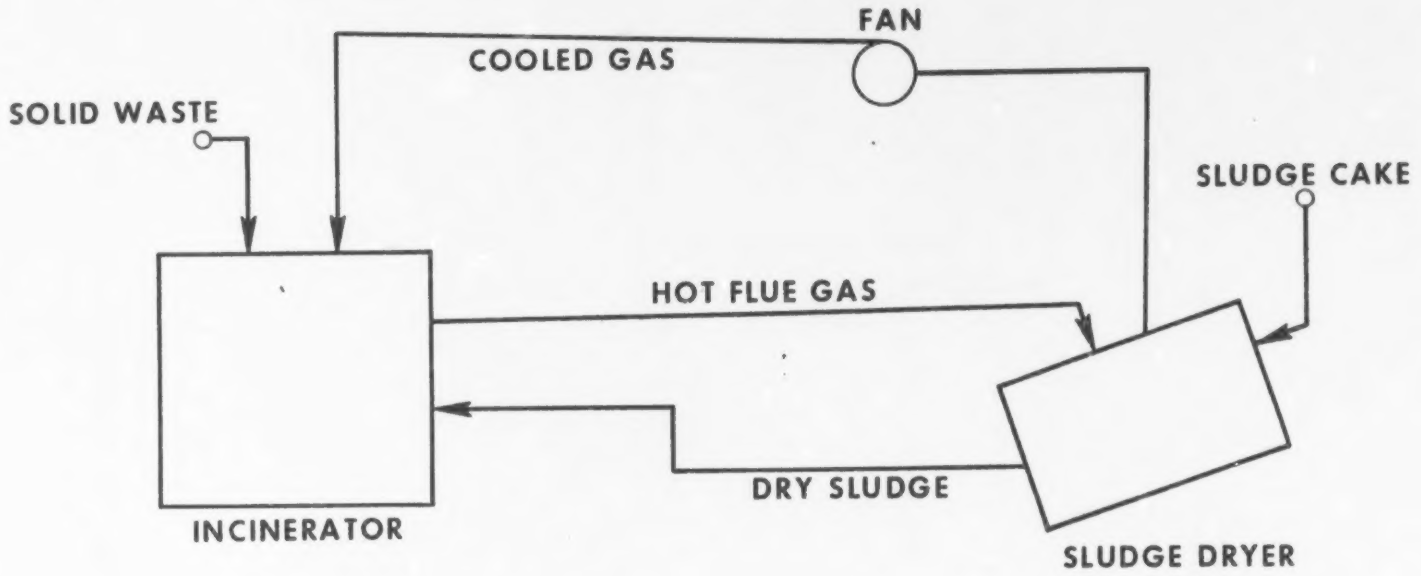

0.2 TONS OXYGEN
1.0 TONS REFUSE
FURNACE
1.01 TONS GAS
GAS CLEANING TRAIN
0.7 TONS FUEL GAS
WASTEWATER 0.28 TONS
0.22 TONS GLASS AND METAL
0.03 TONS RECYCLE

FIGURE 15. SIMPLIFIED SYSTEM FOR COMBINED INCINERATION


FAN
COOLED GAS
SOLID WASTE
SLUDGE CAKE
HOT FLUE GAS
DRY SLUDGE
INCINERATOR
SLUDGE DRYER

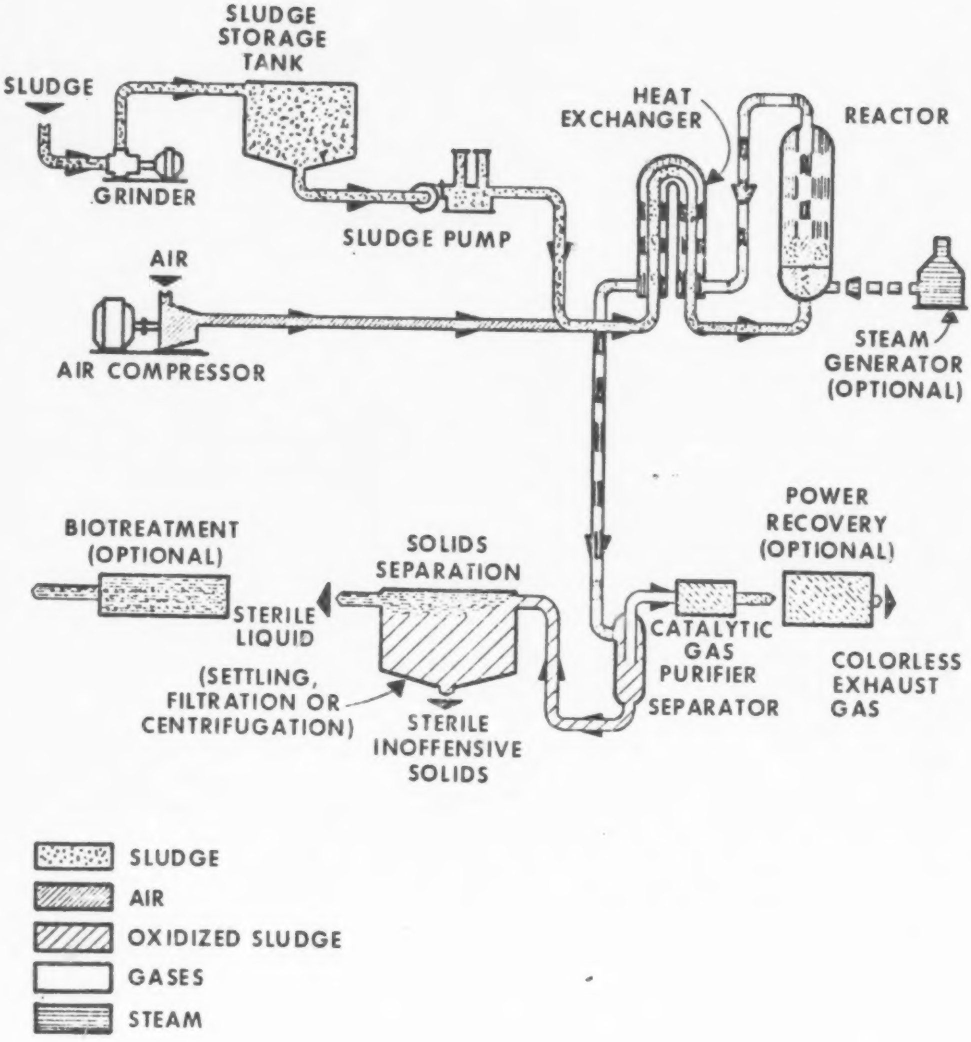

SLUDGE
SLUDGE STORAGE TANK
GRINDER
SLUDGE PUMP
HEAT EXCHANGER
REACTOR
AIR
AIR COMPRESSOR
STEAM GENERATOR (OPTIONAL)
BIOTREATMENT (OPTIONAL)
SOLIDS SEPARATION
STERILE LIQUID
(SETTLING, FILTRATION OR CENTRIFUGATION)
STERILE INOFFENSIVE SOLIDS
CATALYTIC GAS PURIFIER
SEPARATOR
POWER RECOVERY (OPTIONAL)
COLORLESS EXHAUST GAS
SLUDGE
AIR
OXIDIZED SLUDGE
GASES
STEAM

FIGURE 16.
FLOW DIAGRAM FOR CONTINUOUS WET AIR OXIDATION

FIGURE 1. WASTEWATER TREATMENT PLANT

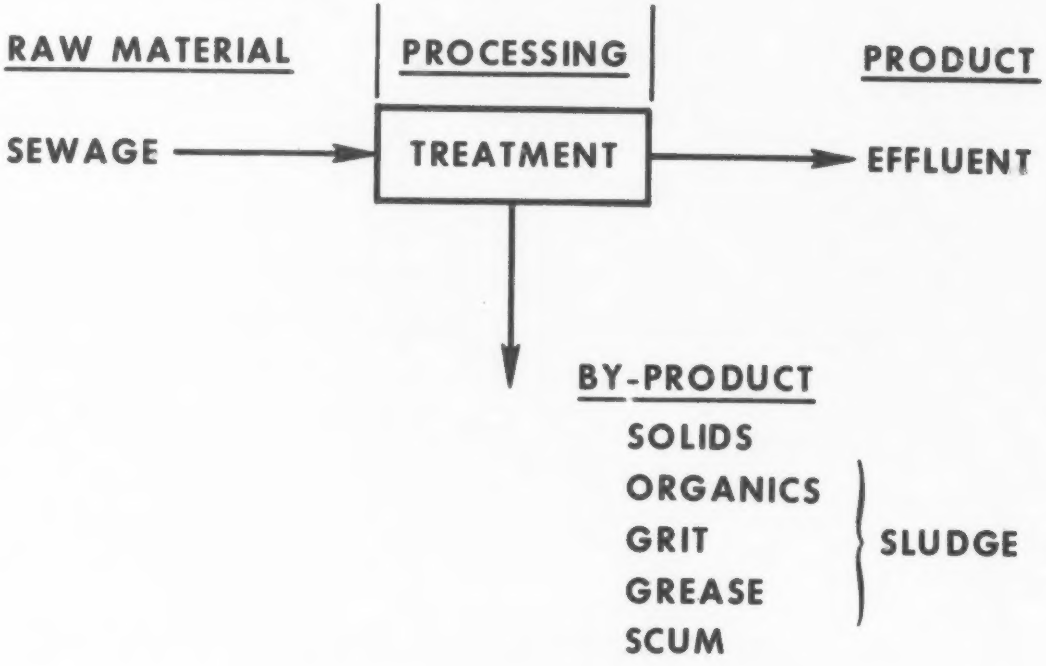
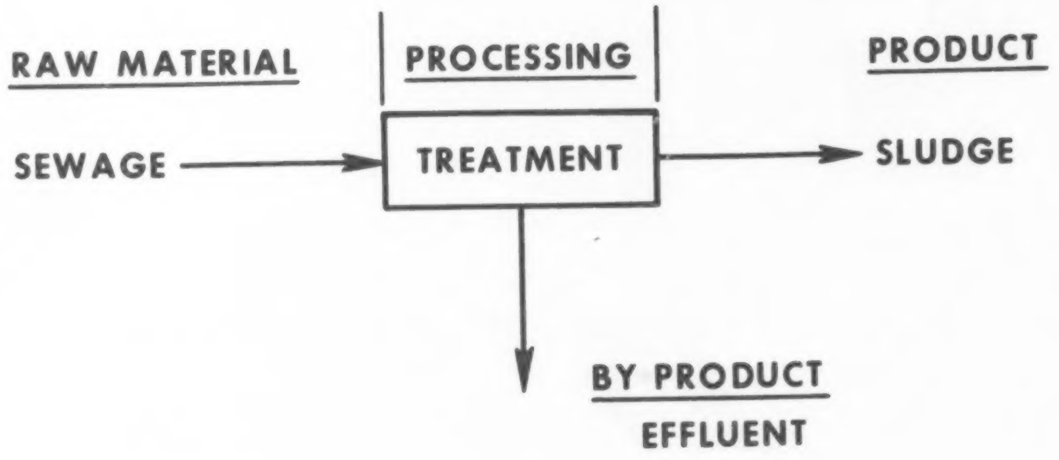

RAW MATERIAL
PROCESSING
PRODUCT
SEWAGE
TREATMENT
EFFLUENT
BY-PRODUCT
SOLIDS
ORGANICS
GRIT
GREASE
SCUM
SLUDGE

FIGURE 2. SEWAGE SLUDGE FACTORY


RAW MATERIAL
PROCESSING
PRODUCT
SEWAGE
TREATMENT
SLUDGE
BY PRODUCT
EFFLUENT

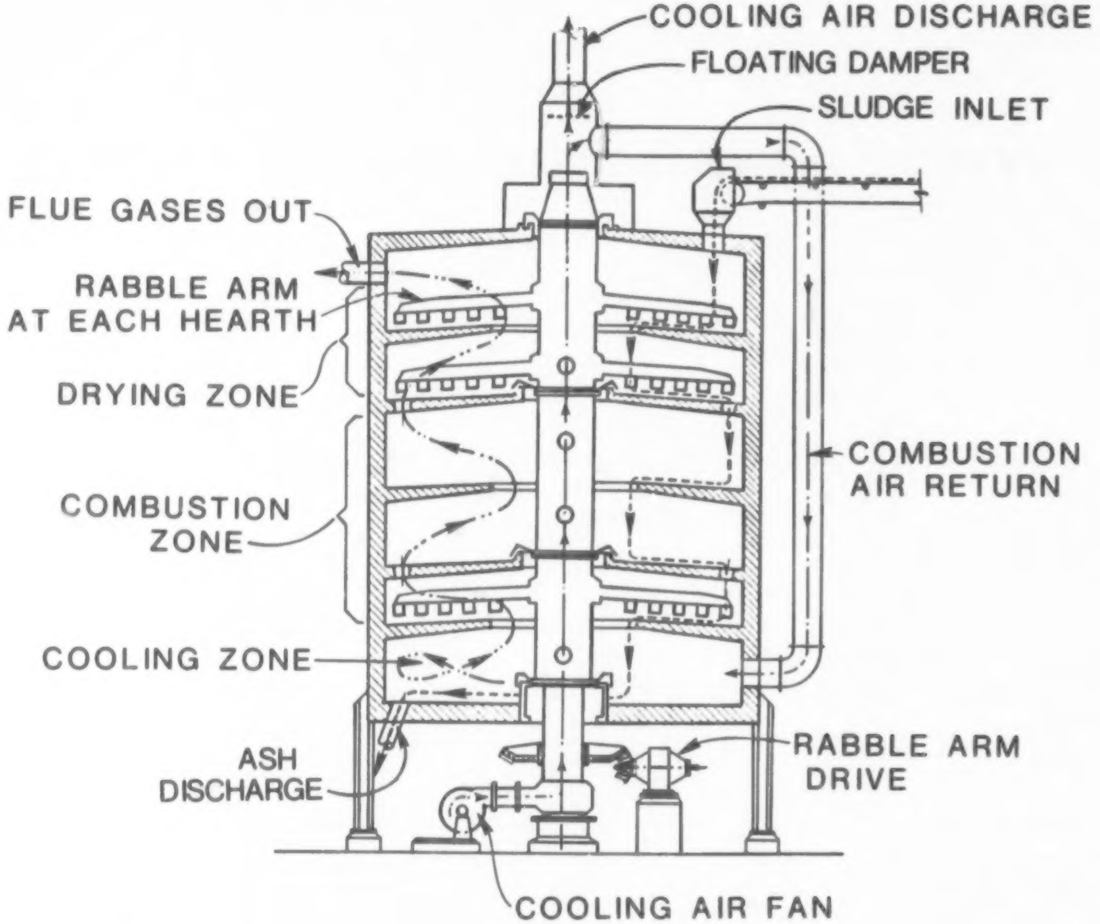
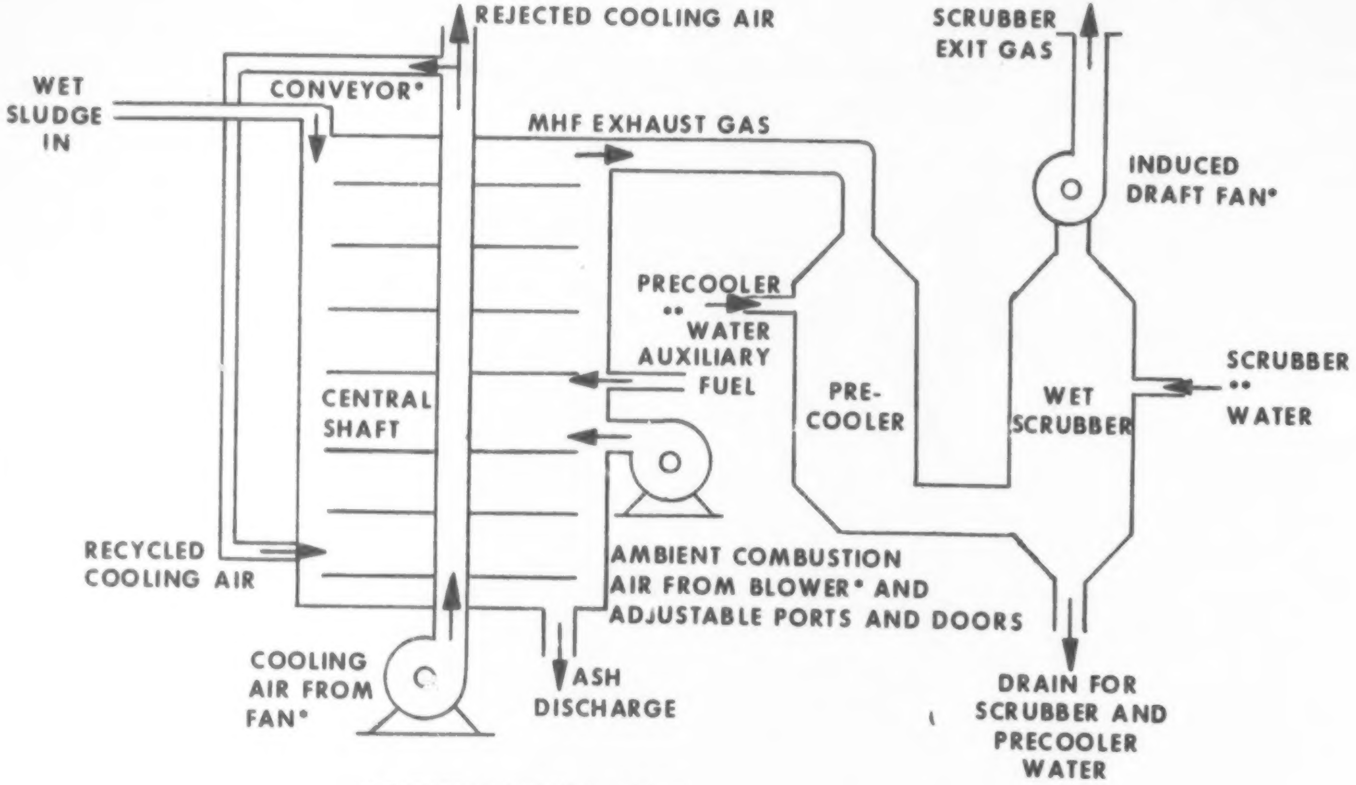

COOLING AIR DISCHARGE
FLOATING DAMPER
SLUDGE INLET
FLUE GASES OUT
RABBLE ARM AT EACH HEARTH
DRYING ZONE
COMBUSTION AIR RETURN
COMBUSTION ZONE
COOLING ZONE
ASH DISCHARGE
RABBLE ARM DRIVE
COOLING AIR FAN

FIGURE 3. TYPICAL SECTION: MULTIPLE HEARTH INCINERATOR


REJECTED COOLING AIR
SCRUBBER EXIT GAS
WET SLUDGE IN
CONVEYOR*
MHF EXHAUST GAS
INDUCED DRAFT FAN*
PRECOOLER ** WATER
AUXILIARY FUEL
CENTRAL SHAFT
PRE-COOLER
WET SCRUBBER
SCRUBBER ** WATER
RECYCLED COOLING AIR
AMBIENT COMBUSTION AIR FROM BLOWER* AND ADJUSTABLE PORTS AND DOORS
COOLING AIR FROM FAN*
ASH DISCHARGE
DRAIN FOR SCRUBBER AND PRECOOLER WATER

* ELECTRIC MOTOR DRIVE
** PUMP WITH MOTOR DRIVE

FIGURE 4. MHF FLOW AND EQUIPMENT DIAGRAM

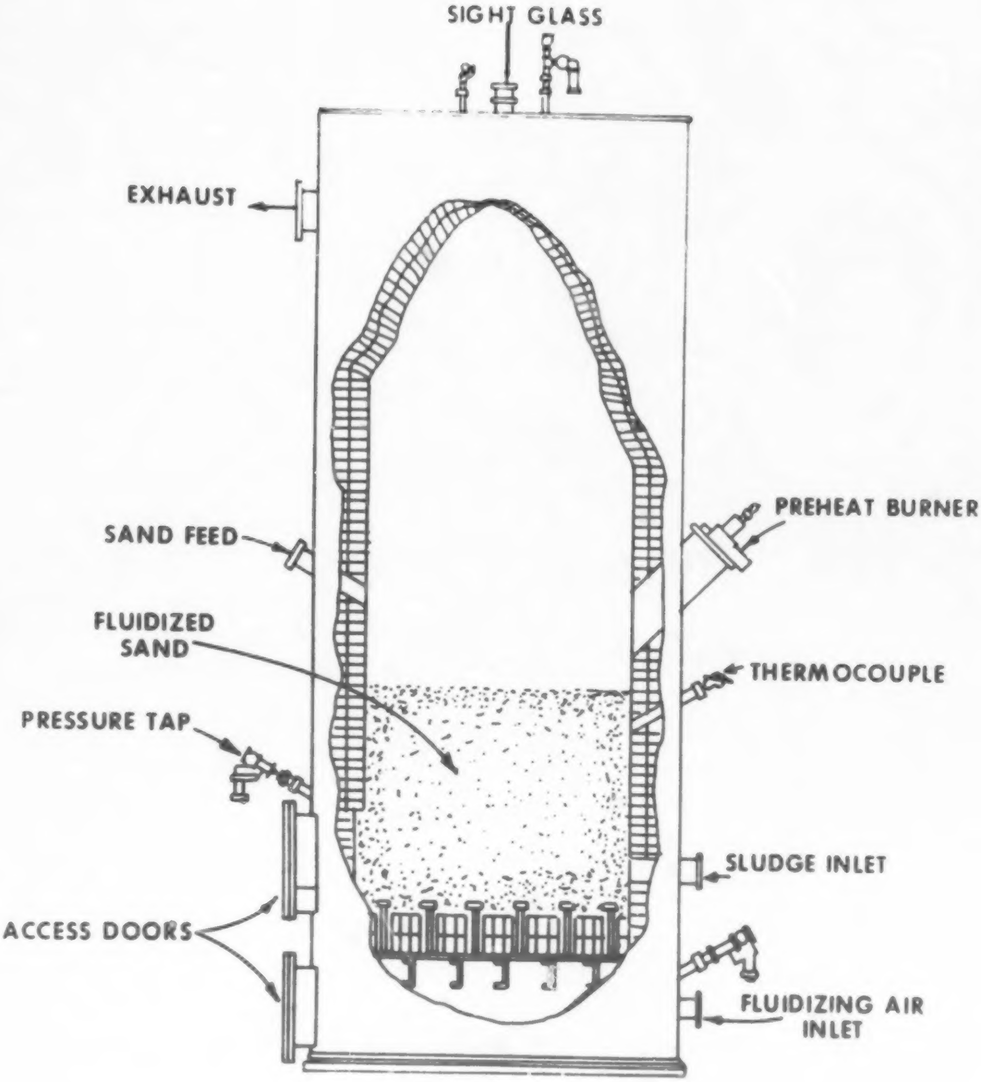

SIGHT GLASS
EXHAUST
SAND FEED
PREHEAT BURNER
FLUIDIZED SAND
THERMOCOUPLE
PRESSURE TAP
SLUDGE INLET
ACCESS DOORS
FLUIDIZING AIR INLET

FIGURE 5.

TYPICAL SECTION OF A FLUID BED REACTOR
(DORR-OLIVER, INC.)

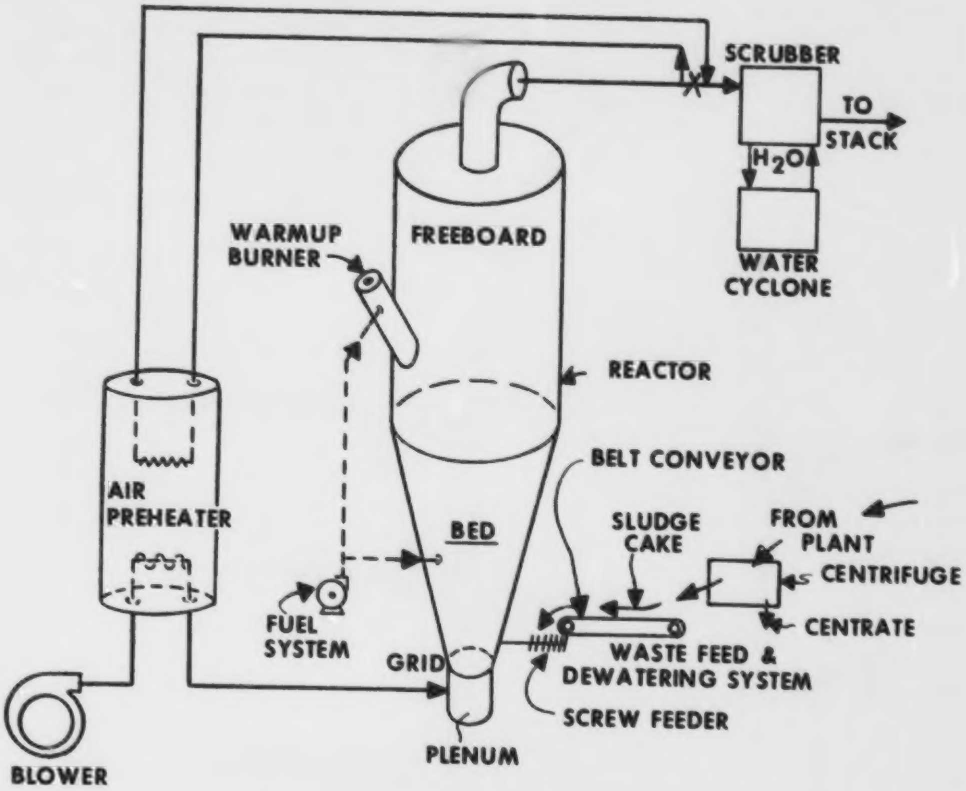
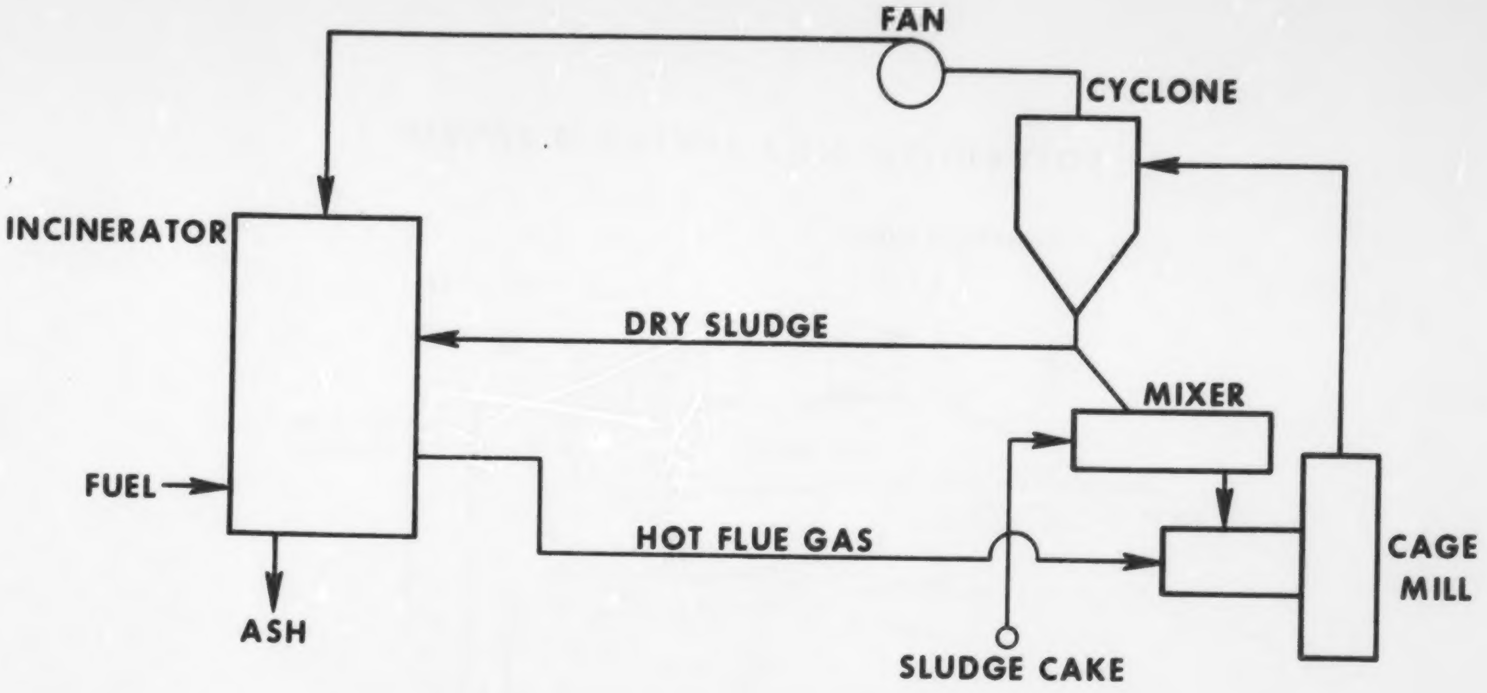

SCRUBBER
TO STACK
H_2O
WATER CYCLONE
WARMUP BURNER
FREEBOARD
REACTOR
BELT CONVEYOR
AIR PREHEATER
BED
SLUDGE CAKE
FROM PLANT
CENTRIFUGE
FUEL SYSTEM
CENTRATE
GRID
WASTE FEED & DEWATERING SYSTEM
SCREW FEEDER
PLENUM
BLOWER

FIGURE 6.
SYSTEM COMPONENTS FOR FB REACTOR

FIGURE 7. FLASH DRYER-INCINERATOR


FAN
CYCLONE
INCINERATOR
DRY SLUDGE
MIXER
FUEL
HOT FLUE GAS
CAGE
MILL
ASH
SLUDGE CAKE

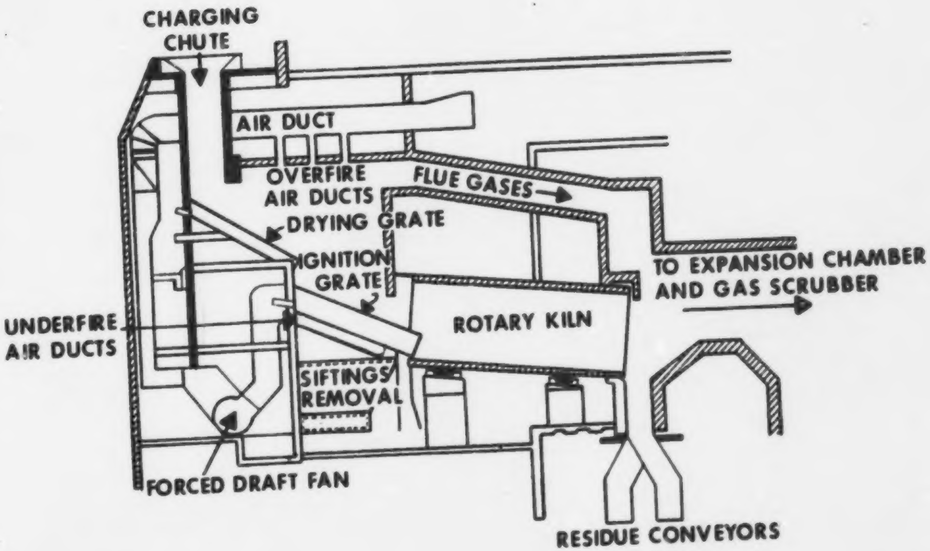

CHARGING CHUTE
AIR DUCT
OVERFIRE AIR DUCTS
FLUE GASES
DRYING GRATE
IGNITION GRATE
UNDERFIRE AIR DUCTS
ROTARY KILN
TO EXPANSION CHAMBER AND GAS SCRUBBER
SIFTINGS REMOVAL
FORCED DRAFT FAN
RESIDUE CONVEYORS

FIGURE 8. ROTARY KILN INCINERATOR

FIGURE 9. CYCLONE FURNACE

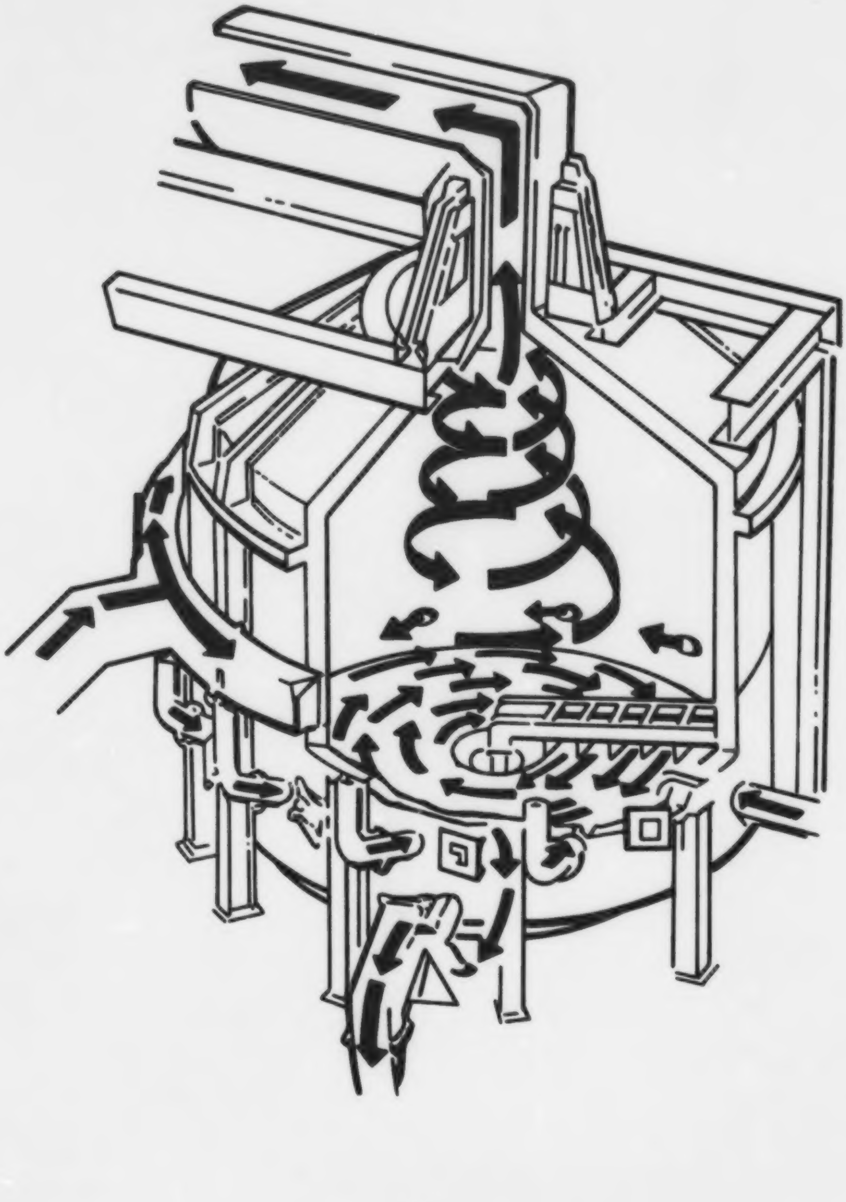


FIGURE 10. BLACK BOX CONCEPT FOR A SLUDGE DISPOSAL SYSTEM

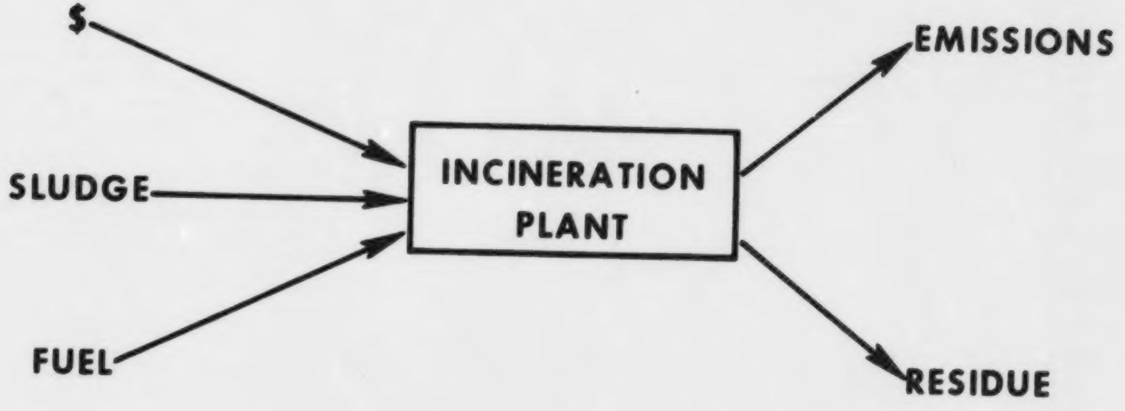
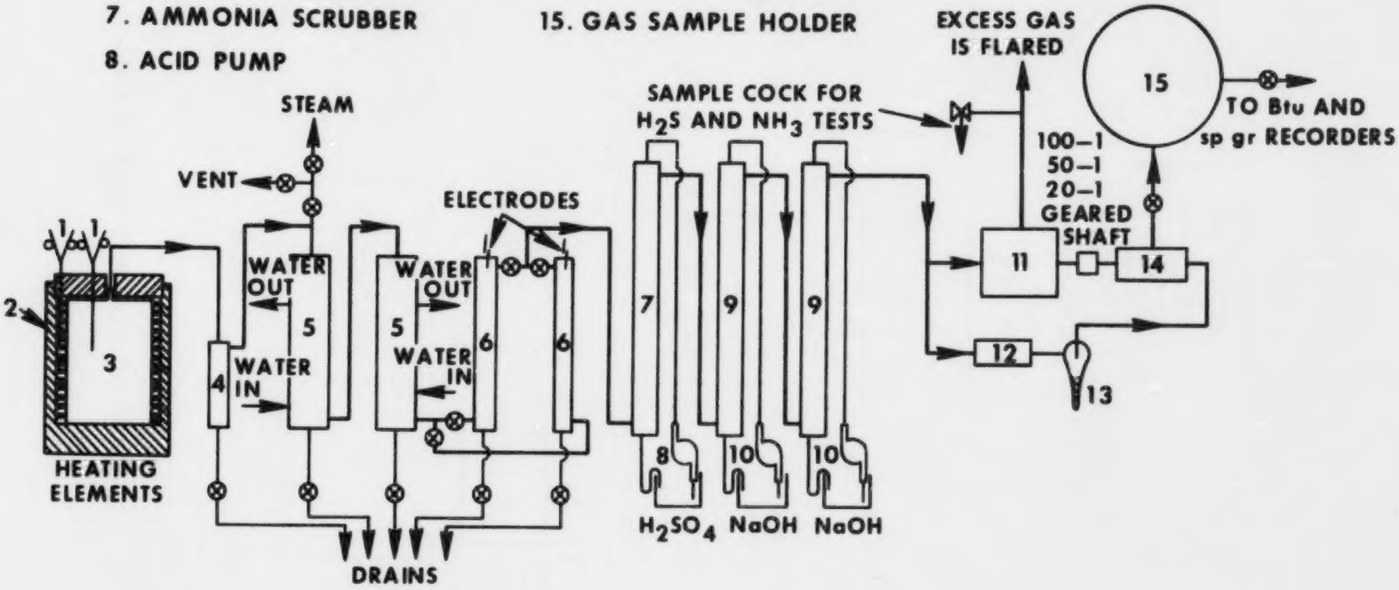

$
SLUDGE
FUEL
INCINERATION PLANT
EMISSIONS
RESIDUE

FIGURE 11. PILOT PLANT PYROLYSIS SYSTEM

LEGEND

1. THERMOCOUPLE
2. ELECTRIC FURNACE
3. RETORT
4. TAR TRAP
5. TUBULAR CONDENSER
6. ELECTROSTATIC PRECIPITATOR
7. AMMONIA SCRUBBER
8. ACID PUMP
9. CARBON DIOXIDE SCRUBBER
10. CAUSTIC PUMP
11. LARGE WET-TEST METER
12. DRYING TUBE
13. LIGHT OIL CONDENSER
14. SMALL WET-TEST METER
15. GAS SAMPLE HOLDER


EXCESS GAS
IS FLARED
15
TO Btu AND
sp gr RECORDERS
SAMPLE COCK FOR
H_2S AND NH_3 TESTS
100—1
50—1
20—1
GEARED
SHAFT
STEAM
VENT
ELECTRODES
1
1
2
3
4
WATER
OUT
5
WATER
IN
5
WATER
OUT
WATER
IN
6
6
7
9
9
8
10
10
11
14
12
13
HEATING
ELEMENTS
DRAINS
H_2SO_4
NaOH
NaOH

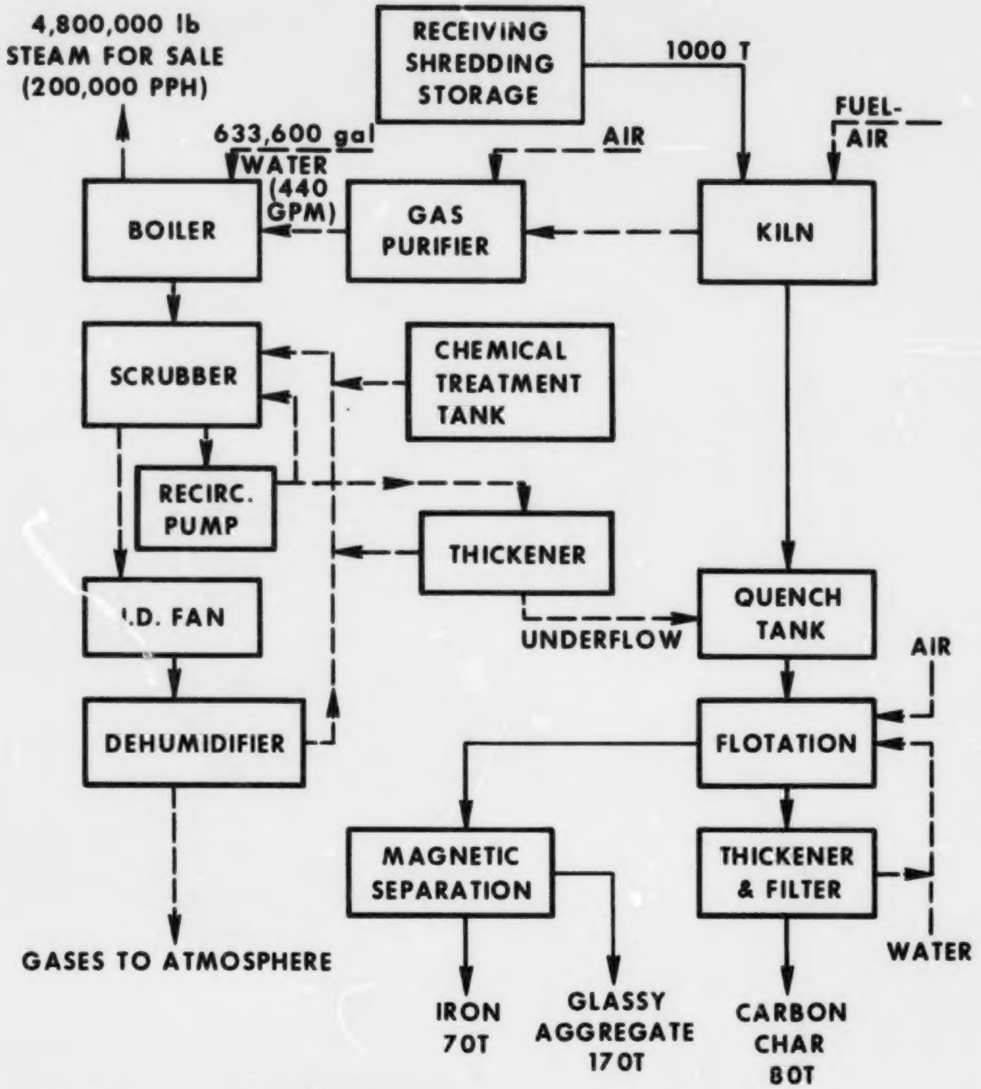

4,800,000 lb
STEAM FOR SALE
(200,000 PPH)
RECEIVING
SHREDDING
STORAGE
1000 T
633,600 gal
WATER
(440
GPM)
AIR
FUEL-
AIR
BOILER
GAS
PURIFIER
KILN
SCRUBBER
CHEMICAL
TREATMENT
TANK
RECIRC.
PUMP
THICKENER
I.D. FAN
UNDERFLOW
QUENCH
TANK
AIR
DEHUMIDIFIER
FLOTATION
MAGNETIC
SEPARATION
THICKENER
& FILTER
GASES TO ATMOSPHERE
WATER
IRON
70T
GLASSY
AGGREGATE
170T
CARBON
CHAR
80T

FIGURE 12. LANDGARD FLOW SHEET; BASIS - 1000 TONS FEED

FIGURE 13. BLOCK DIAGRAM OF THE GR&D SOLID WASTE RECYCLING PROCESS

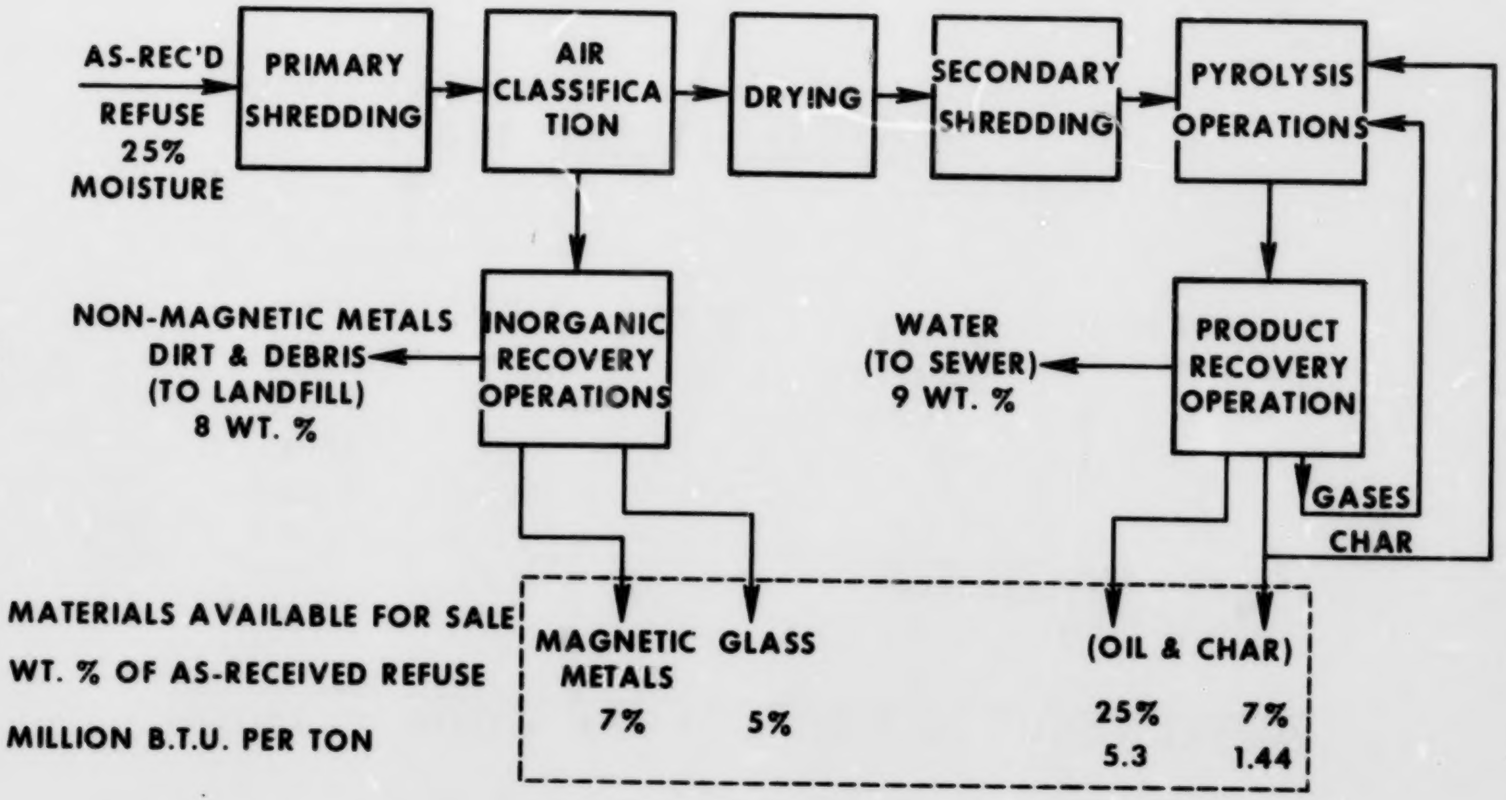

AS-REC'D
REFUSE
25%
MOISTURE
PRIMARY SHREDDING
AIR CLASSIFICA TION
DRYING
SECONDARY SHREDDING
PYROLYSIS OPERATIONS
INORGANIC RECOVERY OPERATIONS
NON-MAGNETIC METALS
DIRT & DEBRIS
(TO LANDFILL)
8 WT. %
PRODUCT RECOVERY OPERATION
WATER
(TO SEWER)
9 WT. %
GASES
CHAR
MATERIALS AVAILABLE FOR SALE
WT. % OF AS-RECEIVED REFUSE
MILLION B.T.U. PER TON
MAGNETIC METALS
7%
GLASS
5%
(OIL & CHAR)
25%
7%
5.3
1.44

FIGURE 14. INPUTS AND PRODUCTS OF PUROX SYSTEM

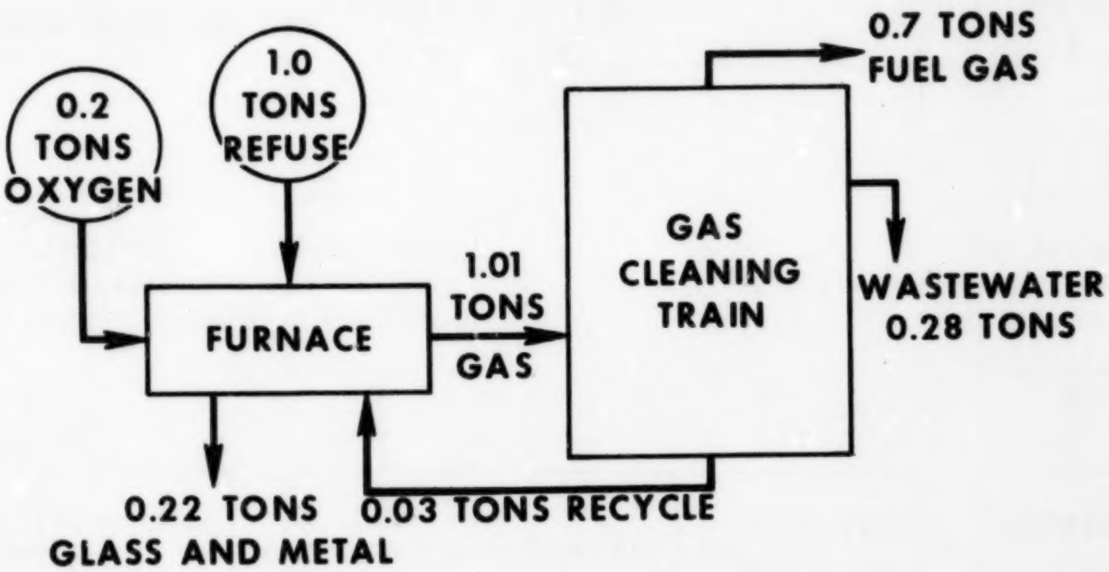
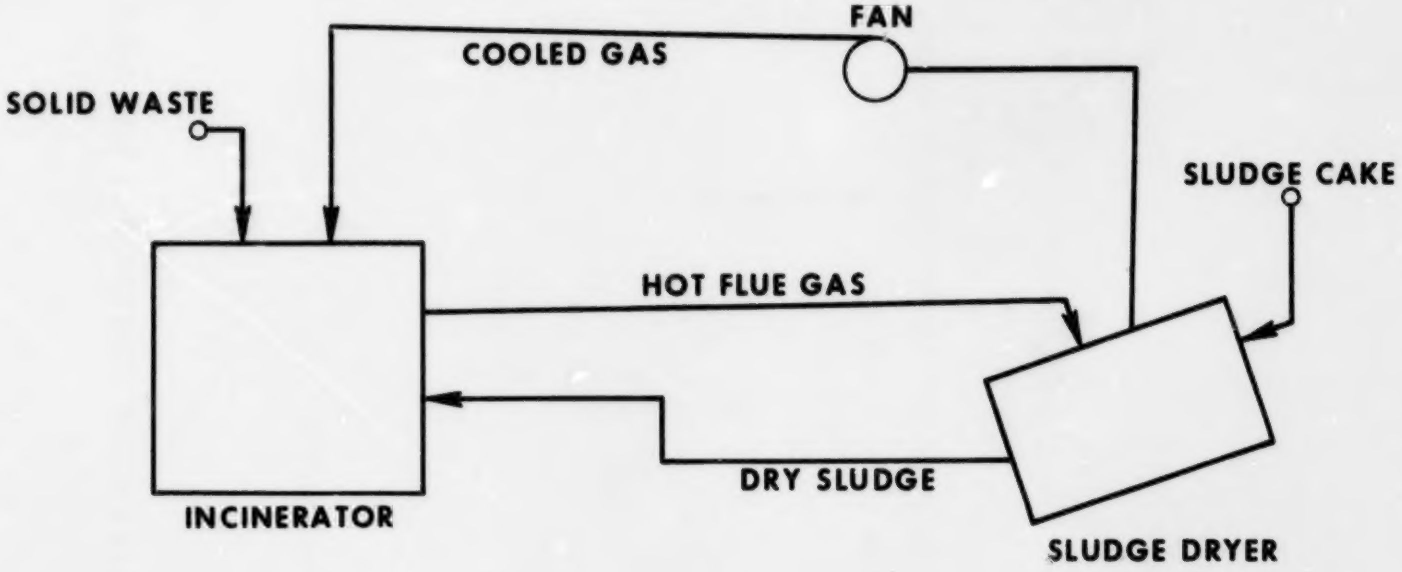

0.2
TONS
OXYGEN
1.0
TONS
REFUSE
FURNACE
1.01
TONS
GAS
GAS
CLEANING
TRAIN
0.7 TONS
FUEL GAS
WASTEWATER
0.28 TONS
0.22 TONS
GLASS AND METAL
0.03 TONS RECYCLE

FIGURE 15. SIMPLIFIED SYSTEM FOR COMBINED INCINERATION


FAN
COOLED GAS
SOLID WASTE
SLUDGE CAKE
HOT FLUE GAS
DRY SLUDGE
INCINERATOR
SLUDGE DRYER

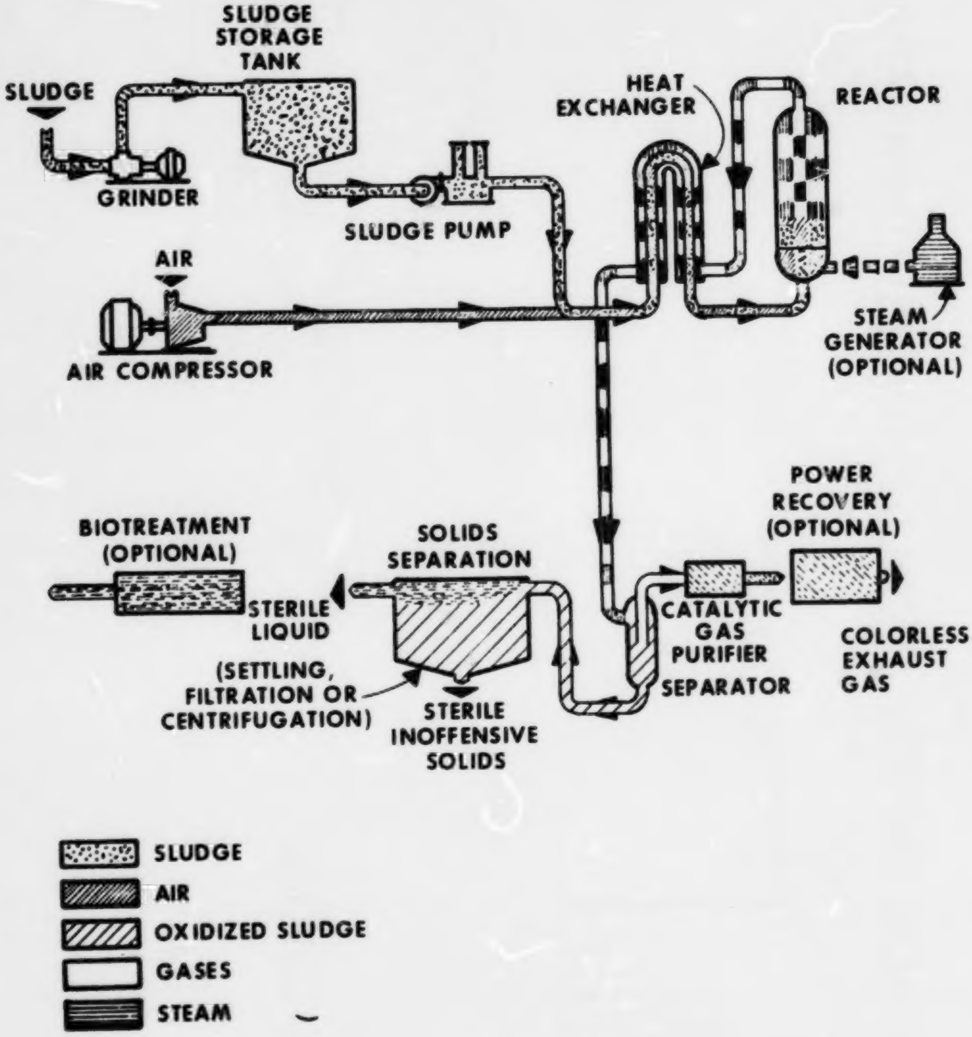

SLUDGE
SLUDGE STORAGE TANK
GRINDER
SLUDGE PUMP
HEAT EXCHANGER
REACTOR
AIR
AIR COMPRESSOR
STEAM GENERATOR (OPTIONAL)
POWER RECOVERY (OPTIONAL)
BIOTREATMENT (OPTIONAL)
SOLIDS SEPARATION
STERILE LIQUID
(SETTLING, FILTRATION OR CENTRIFUGATION)
STERILE INOFFENSIVE SOLIDS
CATALYTIC GAS PURIFIER
SEPARATOR
COLORLESS EXHAUST GAS
SLUDGE
AIR
OXIDIZED SLUDGE
GASES
STEAM

FIGURE 16.
FLOW DIAGRAM FOR CONTINUOUS WET AIR OXIDATION

THICKENING CHARACTERISTICS OF ALUMINUM AND IRON PRIMARY SEWAGE SLUDGES *

by

Steven W. Hathaway and Joseph B. Farrell **

INTRODUCTION

The need for phosphate removal to control eutrophication in certain receiving waters has been well documented (1). In virtually all cases phosphate removal is accomplished by addition of chemical precipitants such as lime and Al or Fe salts. The effectiveness of phosphate removal can be accurately predicted and reliable methods are available for design of the water-handling facilities (1). However, there is insufficient information on which to base the design of equipment for processing the sludges produced by phosphate removal processes. This is particularly true for processes in which Al or Fe salts are used as the precipitating agents.

Average quantities of sludge produced and limited information on dewatering of the sludges produced by phosphate removal processes have been presented by Adrian and Smith (2). Burns and Shell (3) have reported on pilot plant investigations during which phosphate was precipitated by addition of lime, alum, and ferric chloride to a reactor-clarifier. Their results were incomplete, but further work is in progress. Van Fleet et al (4), in what is probably the most authoritative publication to date, described the properties of sludge produced in plant-scale tests of phosphate removal processes in the Province of Ontario, Canada. Nevertheless, there is a dearth of information on dewatering rates obtainable with most dewatering devices and on the effect of process variables on performance.

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One clear indication from the limited information available is that primary sludges which result when Al or Fe salts are added to the primary clarifier (referred to herein as Al-primary or Fe-primary sludges) do not thicken to as high a solids content as do conventional primary sludges. Vacuum filter yields are low, probably at least partially because the sludge to be filtered has a low solids content. The interdependence between thickening and dewatering indicated that thickening characteristics should be explored before investigating performance of dewatering devices. The objective of this investigation, then, is to define the thickening characteristics of Al-primary and Fe-primary sludges by means of standard gravity thickening and air flotation measurements.

MATERIALS AND METHODS

Duplicate Treatment Units--To compare the thickening characteristics of the chemical sludges it was necessary to produce a consistent and reliable source of sludge. Depending on the raw wastewater entering the plant, the sludge produced in a pilot unit may change its character hourly. Therefore, duplicate units were set up to produce two types of chemical sludge under identical flow and equipment design conditions.

Figure 1 represents the treatment system used in this study. It consists of two duplicate primary clarification units that receive the same sewage feed. The duplicate system allowed flexibility of operation:

each unit could be run at different dose levels of the same chemical, or with different chemicals; or one unit could be dosed with chemical and the other with no chemical (run as a conventional primary treatment plant). Each unit was fed raw macerated sewage at a rate of 20 liters per min. (5 gpm) into the chemical mixing and coagulation tank. The chemically-treated sewage then overflowed to the primary clarifier. The lower section of the clarifier was conically shaped to facilitate sludge removal. The overflow rate was set at a low value, 20 m^3/m^2-day (500 gallons/ft^2-day), so that the sludge level could be maintained below the weir. Sludge was withdrawn once daily from each clarifier and stored for the thickening experiments in a holding tank.

The table below shows the target metal to phosphorus atomic ratios (M/P) and the levels at which the chemicals, alum ($Al_2(SO_4)_3 \cdot 14H_2O$) and ferric chloride ($FeCl_3$), were charged to the wastewater to produce the sludges required in the investigation. Chemical dose levels are based on the average P level of 8.0 mg/l. An anionic polymer (2 A 2, Atlas Chemical Co.*) was used only when the units were dosed with chemical.

	Polymer (mg/l)	Al (mg/l)	Fe (mg/l
Control	0	0	0
M/P = 1.2	0.5	8.4	17.1
M/P = 2.0	0.5	14.0	27.0

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* Mention of a trade name or manufacturer does not indicate EPA endorsement.

Gravity Thickening--Gravity thickening of the chemical sludges was studied in bench-top thickening tests using two-liter cylinders as settling columns. Each cylinder was equipped with a three-pronged vertical gate stirrer which rotated at 1 rpm. This rotation was equivalent to a peripheral tip speed of 25 centimeters per minute. Sludge was initially placed in the cylinder to the two-liter mark and, with the gate rotating, allowed to settle for 24 hours. If a polymer was to be used, it was added to the sludge in the cylinder before the settling test and mixed by inverting the cylinder several times. Readings of the level of interface between sludge and supernatant were taken at short intervals of about 10 minutes for a period of about four hours. A final reading was taken at the 24-hour period.

The method of determining the gravity thickening characteristics of a sludge utilizes a graphical method illustrated in Figures 2, 3, and 4. The procedure followed is essentially the same as that outlined by Dick (5), except that the settling velocity vs. sludge concentration curve is obtained from one instead of several settling tests. Gravity-thickening characteristics for each sludge tested were defined by a single settling test. Interface level as a function of time obtained in the settling test were plotted as shown in Figure 2. At any point along this curve, settling velocity can be calculated from the slope, which is estimated from a tangent line. Average concentration of solids in the settling sludge can be calculated from the original solids concentration and volume and the new volume at that point. The settling velocity can then be plotted against the solids concentration as shown in Figure 3. From this curve, selected points are taken to calculate solids flux and are plotted against solids concentration (Figure 4). The thickening rate (solids flux) at a given thickened sludge concentration is predicted from this curve (Figure 4) by selecting the solids concentration on the abscissa, and drawing an operating line tangent to the flux curve. The intersection of the operating line with the ordinate gives the maximum solids flux through the thickener at the selected thickened-sludge concentration. Dick (5) explains the theoretical basis of the method.

Dissolved Air Flotation Thickening--Figure 5 is an illustration of the Komline-Sanderson pilot-scale air flotation thickener used in these experiments. Fresh water was pressurized to 70 psi with air. The pressurized water and chemically-conditioned sludge were then fed into the injection nozzle and into the flotation chamber. The air bubbles entrapped the sludge solids and floated them to the top (6). The thickened sludge was manually scraped into a tank for storage. The underflow (separated liquid) was removed near the tank bottom and exited into the drain. The surface area of the thickener was 0.093 m^2 (1.0 ft^2). The recommended solids loading was 10 kg/m^2-hr (2 lb/ft^2-hr) and the hydraulic loading had to be within 1,500 to 4,000 ml/min. In order to reduce the complexity of the experimentation, the initial solids concentration of the sludge was controlled to about 8,000 mg/l in all experiments by diluting with effluent.

Polymer conditioning of the sludge was essential for all of the flotation tests except for raw primary sludge flotation. The polymer was made up in a 0.1 percent aqueous solution (1 mg/ml) and fed into the sludge feed line just before the sludge entered the flotation unit.

RESULTS AND DISCUSSION

General Observations--The sludges produced by the addition of alum or ferric chloride to the raw sewage were drastically different from those produced by conventional treatment processes. The conventional primary sludge was very fibrous and appeared lumpy. The solid particles were, for the most part, discrete particles suspended in a liquid medium. Although produced from the same sewage, the Al and Fe-primary sludges were not fibrous or lumpy. The chemical sludge was very homogenous with no apparent separation from the liquid phase. An analogy to this observation can be envisioned by comparing the conventional sludge to a chunky vegetable soup and the chemical sludge to an homogenous pea soup. The volume of alum and iron sludges was greater than that of the conventional primary sludge, and the mass was increased by the chemical compounds formed and by the additional suspended solids removed by chemical treatment.

Gravity Thickening Characteristics--The gravity thickening characteristics of primary sludges formed by high and low dose levels of Al and Fe were investigated. The high metal sludges were obtained at a metal to phosphorus ratio of 2.0 and the low metal sludges at a ratio of 1.2. During the experiments, phosphorus (P) and suspended solids (SS) removals were acceptable as is indicated by the average results shown below:

	% P Removal	% SS Removal
High Al	75	64
High Fe	80	74

The higher removals with Fe would tend to produce a poorer settling sludge because more fine solids are present. However, the difference was small so the anticipated effect should not be large enough to invalidate a comparison of the two types of sludges.

Preliminary results showed that both the Al and Fe-primary sludges settled very slowly to low solids content sludges. Exploratory experiments showed that dilution of the sludges with effluent and addition of polymer caused substantial increases in thickening rate (i.e., loading rate of solids per unit area). Consequently, the effect of dilution and polymer addition were investigated in some detail.

The effect of diluting the high Al and high Fe sludges (i.e., produced at M/P = 2.0) with effluent is shown in Table 1. Without dilution, thickening rates are extraordinarily low. Dilution with effluent caused an order-of-magnitude change in the rate of thickening of the Al-primary sludge and a modest increase for the Fe-primary sludge. No explanation for this large difference in effect is apparent.

The data shown in Tables 2 and 3 reveal the effect of polymer addition. Even without dilution, polymer addition increases thickening rates for both sludges to useful levels. With dilution of the sludge with effluent as well as polymer addition, the rate increases still more for both the Fe-primary sludge as well as the Al-primary sludge.

Experiments were also conducted with Al and Fe-primary sludges produced at a low level of metal addition (M/P = 1.2). Average phosphorus and suspended solids removals are shown below:

	% P Removal	% SS Removal
Low Al	60	60
Low Fe	70	60

The lower chemical dose, P-removal, and SS-removal would be expected to produce a less mineralized sludge with lower fine-solids content than was obtained at the higher dose (M/P = 2.0). Results of thickening tests are shown in Table 4. Polymer was used in all tests. Surprisingly enough, results do not differ significantly from those obtained with the high metal doses.

A summary of the effects of polymer addition and dilution on the chemical-primary sludges is presented in Table 5. Ranges of solids content and thickening rate are presented. The experimental conditions (a testing period of several months and a high daily variability between different sludges) invalidated somewhat the meaning of comparisons between the results of different treatments. Nevertheless, the following conclusions can be reasonably made, because they are substantiated by significant patterns in the data obtained.

a. Al-primary and Fe-primary sludges have extremely low thickening rates, even when the solids concentration leaving the thickener is low. Conventional gravity thickening appears to be inadequate for these sludges.

b. Dilution of Al-primary sludges with effluent before thickening produces a dramatic increase in the thickening rate and increases thickened solids concentration. This effect of dilution on Fe-primary sludges is substantial but thickening rate is still not brought up to practicable levels.

c. Addition of polymers substantially increases the thickening rate and thickened solids concentrations for both Al and Fe-primary sludges. Rates are increased to practicable levels.

d. When both dilution and polymer addition (b and c above) are used, the thickening rates of Al and Fe-primary sludges are unusually high, well above the rates obtained when polymer alone is used.

e. The measured solids concentration achieved by gravity thickening, even under the best conditions, did not exceed 5 percent for either sludge type. This is a reasonable concentration for a feed to a vacuum filter but is substantially lower than the concentration achieved by thickening conventional primary sludges.

The large effect of polymer addition on the thickening rate achieved in these experiments was not surprising. The use of polymers to improve gravity thickening is well known. The improvement is evidently related to the flocculating effect of the polymer on the sludge particles. Dilution of sludges has been used by others (6) to improve gravity thickening, although the mechanism by which thickening is improved is not clear. The very large improvement in thickening rate with Al-primary sludge was unexpected. No explanation of the poorer performance with Fe-primary sludge is readily apparent.

Air Flotation Thickening--Several different sets of tests were completed using the air flotation thickener for the Al and Fe-primary sludges and also the conventional primary sludge. To have a basis for comparing thickening characteristics, we used conventional primary sludge and Fe-primary sludge in the first set of tests. These sludges were produced by the duplicate physical-chemical units shown in Figure 1. The conventional sludge was generated by settling the sewage in the clarifier without the aid of chemicals. The Fe-primary sludge was generated by adding 27 mg/l Fe^{+3} (M/P = 2.0) and 0.5 mg/l anionic polymer to the sewage. Phosphorus and suspended solids removal are shown below:

	% P Removal	% SS Removal
No chemical	13	52
High Fe	87	72

The lower phosphorus and suspended solids removal obtained with the unit receiving no chemical would be expected to produce a sludge less mineralized and containing a far lower proportion of fine solids than the sludge from the Fe-dosed unit.

Experimental conditions and results of the flotation tests are presented in Table 6. Conventional sludge was fed into the unit at an average of 205 kg/m^2-day (41 lb/ft^2-day) solids loading rate. The sludge readily floated to an average of 7 percent solids with no chemical conditioning. The solids capture was 90 percent or greater. The Fe-primary sludge fed in at approximately the same rate, 180 kg/m^2-day (36 lb/ft^2-day), gave much poorer results. It required anionic polymer conditioning to achieve an average of 3.2 percent final solids. Solids capture ranged from 60 to 83 percent.

The poor flotation characteristics of the Fe-primary sludge is attributed to two factors: the presence of the ferric hydroxide and phosphate, and the high proportion of fine particles in the sludge.

Another set of experiments was carried out to determine the effect of important variables or factors on the solids content to which the sludge could be thickened and on the solids loss to the underflow. The factors selected were metal to phosphorus ratio, dose of polymeric conditioner to the sludge, and feed rate of sludge to the thickener. Each variable was investigated at two levels. The minimum number of experiments for evaluating these three variables or factors at two levels was eight (a 2^3 factorial experiment). Enough sludge for the eight experiments could be collected in two days, but only four experiments could be run in one day. Since sludge properties can change if sludge is stored even for one day, a blocked factorial experiment was designed with the day of testing as the blocking factor. Numerous statistical texts, e.g., Davies et al (7) explain the principles of factorial experimental design, including techniques for blocking out the effects of certain factors (i.e., variables).

The experimental design is shown in Tables 7 and 8: Table 7 shows the levels of the factors investigated, and Table 8 shows the design matrix of the experiment. The four Block 1 tests were run on the first day following collection of sludge and Block 2 tests were run on the second day. The blocking factor was confounded with (confused with) the M x P x L interaction (see footnote to Table 9 for explanation of an interaction). The M x P x L interaction can no longer be calculated, but, assuming that effects of factors are linear, other effects are unaffected.

Phosphorus removals were checked during the periods when sludges were being collected for the factorial experiments and the following results were obtained:

	% P Removal	
	Al	Fe
Low level (M/P = 1.2)	42	57
High level (M/P = 2.0)	45	67

Phosphorus removals were poor when the units were producing Al-primary sludge, because some very light floc escaped capture in the clarifier. Despite losses of solids, the sludge did not appear unusual and did not gravity-thicken to a higher than usual solids content. Removal of phosphorus was better when the units were producing Fe-primary sludges. Better removal tends to produce a more mineralized sludge with more fine particles, which may account in part for the poorer results (see below) obtained with Fe-primary sludge.

An independent estimate of the experimental error in measuring the response, "solids content of the sludge," and "solids loss into the underflow," was obtained by making four successive runs on Fe-primary sludge at constant operating conditions. The standard deviations of the response were determined and the 95 percent confidence interval of an "effect" in a 2^3 factorial experiment were calculated:

	Solids Content (%)	Solids Loss (%)
Standard deviation	0.24	2.9
95% confidence interval	± 0.54	± 6.7

The 95% confidence interval for solids loss is excessively large. It means that the experiment cannot establish with reasonable certainty effects of variables on solids loss unless they differ from the average by 6.7 percent. Although the effect of variables on both solids content of the sludge and the solids loss into the underflow were determined in the experiments, the effects of the various factors on solids loss were not reliable and are not reported in detail. Average solids loss for the eight tests making up the factorial experiment was 13.0 percent for the Al-primary sludge and 8.1 percent for the Fe-primary sludge.

The effect of variables on the solids content of the sludge calculated from the experimental results of the factorial experiments are presented in Table 9. Effects in the table marked with an asterisk were significant. The effects of the main factors--mineral dose, polymer dose, and solids loading--were all significant. Although the magnitude of their effects differ, they each affected sludge solids content in the same direction. The average solids content was higher for the Al-primary sludges than for the Fe-primary sludges. No significance should be given to this observation because the sludges were collected at different times.

The results of the experiments at the high solids loading are presented graphically in Figure 6. The data points have been calculated from the averages and significant effects shown in Table 9. For both the Al-primary and Fe-primary sludges, increased metal to phosphorus ratio reduced the solids content of the sludge. Polymer dose increased solids content for both types of sludge.

SUMMARY AND RECOMMENDATIONS

The experiments demonstrate that Al-primary and Fe-primary sludges do not gravity thicken as well as conventional primary sludges. Al and Fe-primary sludges produced from wastewater at Lebanon, Ohio, gravity thicken to about 4-5 percent solids. Bench-scale experiments have shown that rates can be greatly improved by dilution with effluent or polymer conditioning or both. Demonstration that dilution and/or polymer conditioning improves thickening in continuously operated equipment is needed.

Air flotation is a satisfactory means for thickening Al and Fe-primary sludges. Performance is not as good as with conventional primary sludge but solids content of 5-8 percent can be achieved, with solids recovery in excess of 90 percent. Anionic polymer doses of 4-8 lb/ton dry solids improves performance. Poorer results are obtained at higher doses of metal to the wastewater.

Only a limited number of polymeric flocculants were examined in this study. There is good likelihood that a more comprehensive screening would uncover polymers which produce higher solids content sludges at higher thickening rates.

Preliminary filtration experiments carried out at Lebanon have shown that filtration of Al and Fe-primary sludges can be carried out at reasonable rates, provided the sludge solids content is 4 percent or above. This solids content can be achieved by gravity or flotation thickening. Consequently, this investigation has demonstrated that Al and Fe-primary sludges can be thickened to concentrations that make subsequent dewatering steps practicable.

Thickening rates are sufficiently high so that capital costs for equipment will not be excessive. When polymeric conditioning was needed, dose levels were not excessive so chemical costs would be reasonable.

ACKNOWLEDGEMENTS

The assistance of J. T. Burden and O. L. Grant in carrying out these investigations is gratefully acknowledged.

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TABLE 1

EFFECT OF DILUTION ON GRAVITY THICKENING OF HIGH Al AND Fe PRIMARY SLUDGES

(NO POLYMER CONDITIONING)

Sludge Type	Initial Solids %	Dilution With Effluent	Calculated Solids Loading lb/ft^2-day	Thickened Solids % At Calculated Solids Loading
High Alum	1.7	None	< 0.4	< 2.0
High Alum	1.2	None	3.4	3.4
High Alum	0.9	1:1	20.0	4.0
High Alum	1.5	1:1	38.0	5.0
High Alum	0.8	1:1	54.0	5.0
High $FeCl_3$	1.2	None	< 0.4	< 2.0
High $FeCl_3$	1.2	None	0.4	3.0
High $FeCl_3$	1.2	1:1	1.5	4.5
High $FeCl_3$	0.9	1:1	3.0	4.0

TABLE 2

GRAVITY THICKENING OF HIGH DOSE ALUM PRIMARY SLUDGE

(ANIONIC POLYMER ADDED FOR SLUDGE CONDITIONING)

Sludge Type	Initial Solids %	Dilution With Effluent	Chemical Conditioning lb Polymer/Ton	Calculated Solids Loading lb/ft^2-day	Thickened Solids % At Calculated Solids Loading
High Alum	1.8	None	6.0	90	5.0
High Alum	1.2	None	6.0	21	4.0
High Alum	1.3	None	6.0	95	5.0
High Alum	1.7	None	4.0	60	4.0
High Alum	0.8	1:1	4.0	139	4.0
High Alum	0.9	1:1	6.0	367	3.0

TABLE 3

GRAVITY THICKENING OF HIGH DOSE IRON PRIMARY SLUDGE

(ANIONIC POLYMER ADDED FOR SLUDGE CONDITIONING)

Sludge Type	Initial Solids %	Dilution With Effluent	Chemical Conditioning lb Polymer/Ton	Calculated Solids Loading lb/ft^2-day	Thickened Solids % At Calculated Solids Loading
High $FeCl_3$	1.3	None	4.0	17.0	4.0
High $FeCl_3$	1.2	None	3.0	21.0	4.0
High $FeCl_3$	1.2	None	2.0	70.0	4.0
High $FeCl_3$	1.3	None	2.0	58.0	4.0
High $FeCl_3$	0.9	1:1	8.0	150.0	3.0
High $FeCl_3$	1.2	1:1	2.0	93.0	5.0
High $FeCl_3$	0.9	1:1	1.8	260.0	3.5

TABLE 4

GRAVITY THICKENING OF LOW DOSE Al AND Fe-PRIMARY SLUDGES: ANIONIC POLYMER ADDED FOR SLUDGE CONDITIONING

Sludge Type	Initial Solids %	Dilution With Effluent	Chemical Conditioning lb Polymer/ Ton D.S.	Calculated Solids Loading lb/ft^2-day	Thickened Solids % At Calculated Solids Loading
. . . .	. . . .	. . . . . . Al-Primary Sludge			
Low Alum	0.90	1:1	1.0	70	5.0
Low Alum	1.25	1:1	6.0	100	5.0
Low Alum	1.24	1:1	2.0	135	5.0
Low Alum	1.30	None	7.0	21	4.0
. . . .	. . . .	. . . . . . Fe-Primary Sludge			
Low Fe	1.2	None	2.0	103	4.0
Low Fe	1.2	None	5.0	43	4.0
Low Fe	1.2	1:1	4.0	135	5.0
Low Fe	1.5	1:1	4.0	450	5.0

TABLE 5

SUMMARY OF EFFECT OF VARIABLES ON THICKENING RATE AND SOLIDS CONTENT OF Al AND Fe-PRIMARY SLUDGES

Sludge Type	Polymer Addition	Dilution	Solids Flux ($lb/ft^2/day$)	% Solids of Thickened Sludge
High Al	No	No	< 0.4-3.4	< 2-3.4
High Al	No	Yes	20-60	4-5
High Fe	No	No	< 0.4	< 2-3
High Fe	No	Yes	1.5-3	4-4.5
High Al	Yes	No	20-90	4-5
High Al	Yes	Yes	140-370	3-4
High Fe	Yes	No	17-70	4
High Fe	Yes	Yes	90-260	3-5
Low Al	Yes	No	21	4
Low Al	Yes	Yes	70-135	5
Low Fe	Yes	No	40-100	4
Low Fe	Yes	Yes	135-450	5

lb/ft^2-day x 4.9 = kg/m^2-day

TABLE 6

AIR FLOTATION THICKENING - CONVENTIONAL PRIMARY - IRON PRIMARY SLUDGE

Sludge Type	Solids Loading Rate lb/ft^2-day	Initial Solids %	Final Solids %	Polymer Dose ** lb/Ton Dry Solids	Solids Capture %
Conventional Primary					
Range	34-48	0.8-1.5	5.1-8.7		
Average	41	1.0	7.4	0	> 90
*$FeCl_3$ Primary					
Range	34-48	0.7-1.3	2.8-3.5	3-6	60-83
Average	36	0.95	3.2		

* High Dose of $FeCl_3$ - 27 mg Fe/l to Sewage
Fe^{+3}/P Molar Ratio = 2.0

** American Cyanamid 837-A

TABLE 7

LEVEL OF FACTORS INVESTIGATED IN 2^3 FACTORIAL EXPERIMENT: Al AND Fe-PRIMARY SLUDGE

Level of Variable	(lb/Poly/Ton Solids) Poly Dose	Mineral Addition M^{+3}/P Molar Ratio	(lb/ft^2 -day) Solids Loading Rate to Air Flotation Unit
- (Low)	1.5	1.2	24
+ (High)	8	2.0	96

TABLE 8

DESIGN MATRIX OF AIR FLOTATION EXPERIMENT WITH Al-PRIMARY AND Fe-PRIMARY SLUDGES

Test	Level of Variables * Metal	Polymer	Loading	M x P x L Interaction	Block
1	-	-	-	-	1
2	+	-	-	+	2
3	-	+	-	+	2
4	+	+	-	-	1
5	-	-	+	+	2
6	+	-	+	-	1
7	-	+	+	-	1
8	+	+	+	+	2

* (-) indicates low level, (+) indicates high level

TABLE 9

EFFECT OF FACTORS AND THEIR INTERACTIONS ON % SLUDGE SOLIDS - AIR FLOTATION OF Al PRIMARY SLUDGE AND Fe PRIMARY SLUDGE

	% Sludge Solids[1]	
	Al	Fe
Average of 8 tests	(6.2)	(5.2)
Mineral dose (M)	- 0.7 *	- 1.6 *
Polymer dose (P)	0.8 *	0.8 *
Solids loading (L)	1.4 *	0.6 *
M x P[2]	- 0.2	0.2
M x L	- 0.7 *	- 0.5
P x L	0.7 *	0.3
M x P x L (Block Effect)[3]	0.1	- 0.4

[1] The average of the results at the low level of the factor or interaction subtracted from the average of the results at the high level.

[2] M x P is a two-factor interaction. M x P = -0.2 means that the effect of P is 0.2 less at the high level of M than at the low level of M.

[3] M x P x L is a three-factor interaction. M x P x L = 0.1 would ordinarily indicate that the M x P interaction is 0.1 more at the high level of P than at the low level. In this experiment M x P x L is "confounded" (or confused) with the block effect so neither can be determined.

* Effects marked with an asterisk are significant.

TABLE 10

EFFECT OF VARIABLES ON % SLUDGE SOLIDS AND % SOLIDS LOSS - AIR FLOTATION OF Fe PRIMARY SLUDGE

	% Sludge Solids *	% Solids Loss *
Average of 8 tests	(5.2)	(8.1)
Mineral Dose (M)	- 1.6	1.3
Polymer Dose (P)	0.8	- 6.2
Solids Loading (L)	0.6	- 2.7
M x P	0.2	- 2.2
M x L	- 0.5	- 6.8
P x L	0.3	0.8
M x P x L (Block Effect)	- 0.4	3.8

* The average of results at the low level of the variable subtracted from the average of results at the high level.

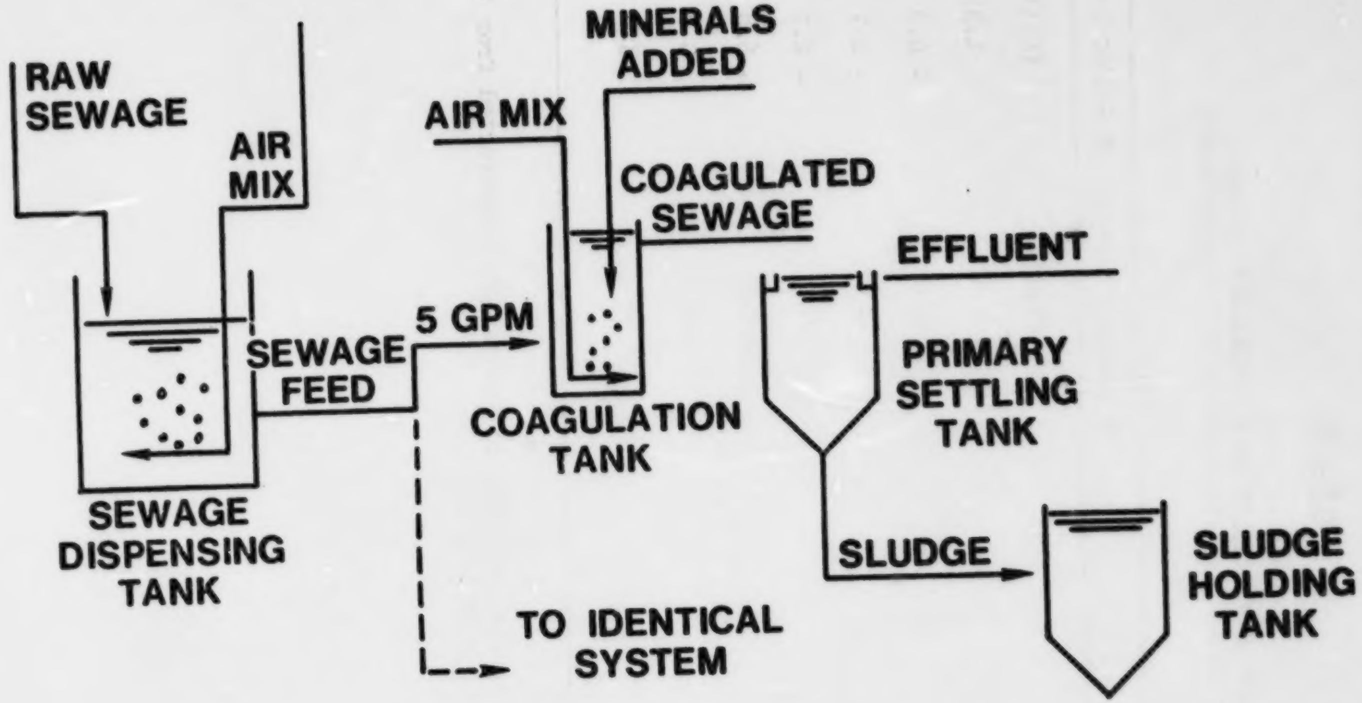
RAW SEWAGE
AIR MIX
SEWAGE FEED
5 GPM
SEWAGE DISPENSING TANK
TO IDENTICAL SYSTEM
AIR MIX
MINERALS ADDED
COAGULATED SEWAGE
COAGULATION TANK
EFFLUENT
PRIMARY SETTLING TANK
SLUDGE
SLUDGE HOLDING TANK

FIGURE 1
DUPLICATE PHYSICAL-CHEMICAL TREATMENT UNITS

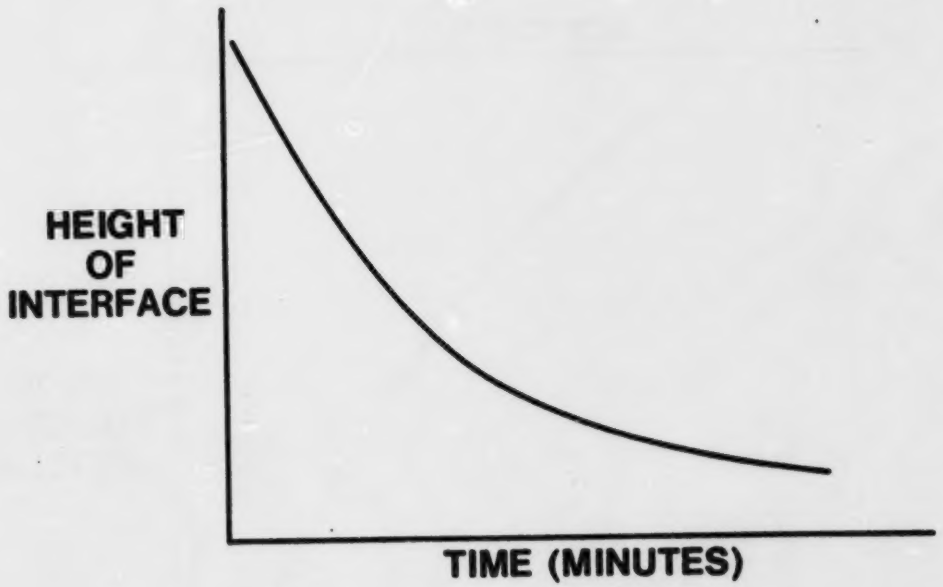

FIGURE 2
STANDARD SETTLING CURVE

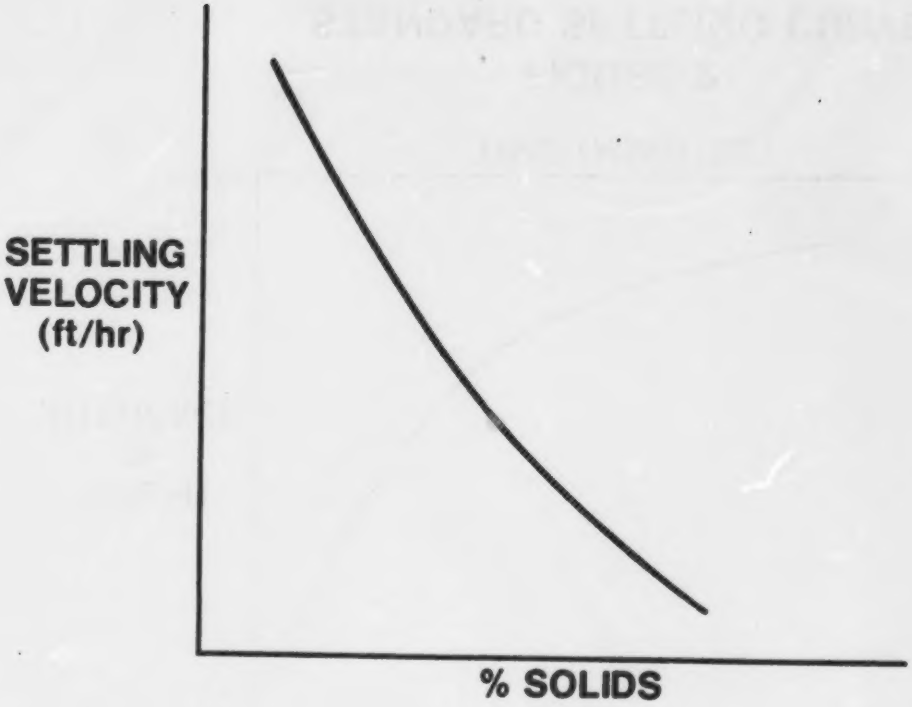

FIGURE 3
SETTLING VELOCITY

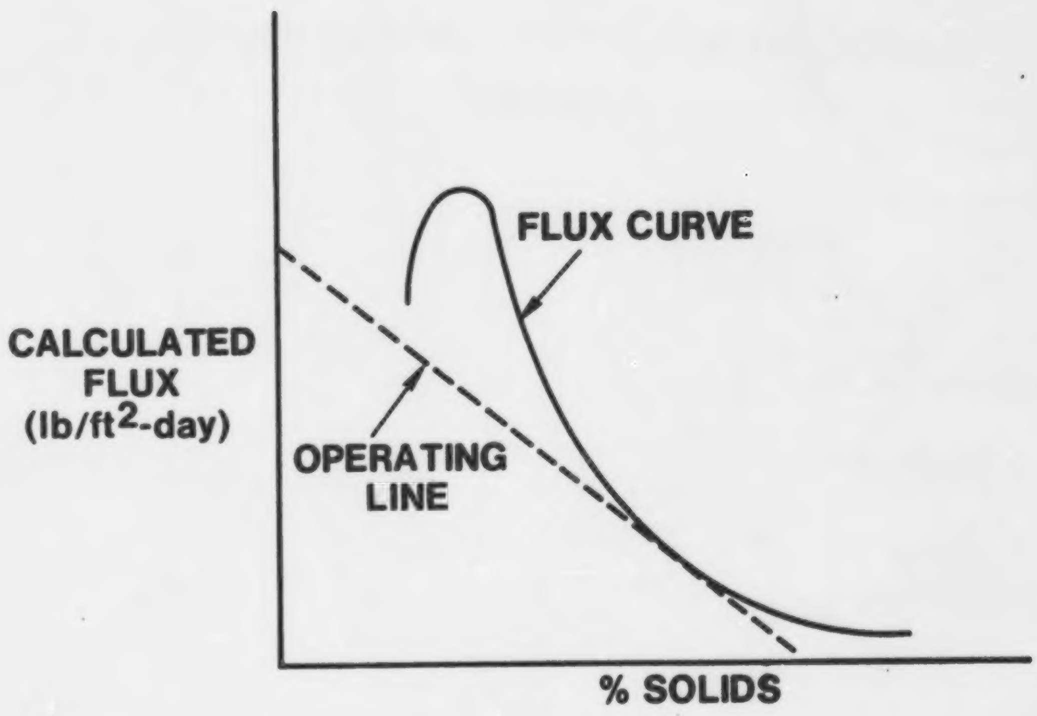

FIGURE 4
FLUX CURVE

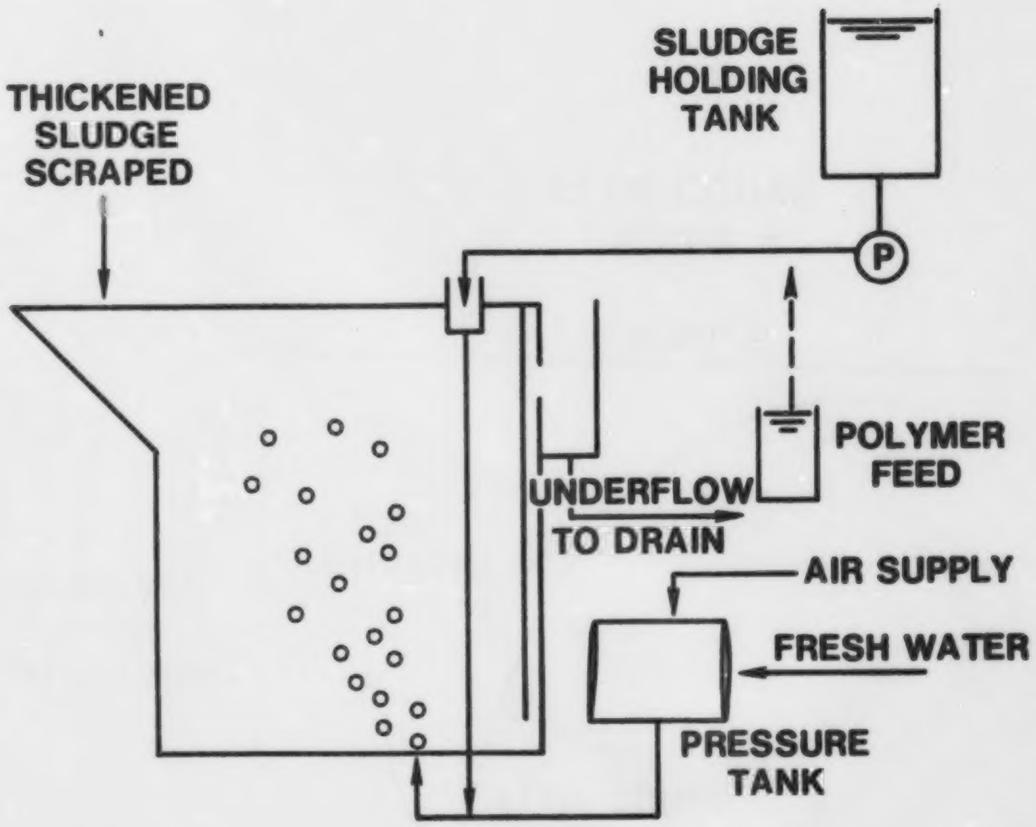
THICKENED SLUDGE SCRAPED
SLUDGE HOLDING TANK
P
POLYMER FEED
UNDERFLOW TO DRAIN
AIR SUPPLY
FRESH WATER
PRESSURE TANK

FIGURE 5
DISSOLVED AIR FLOTATION THICKENER

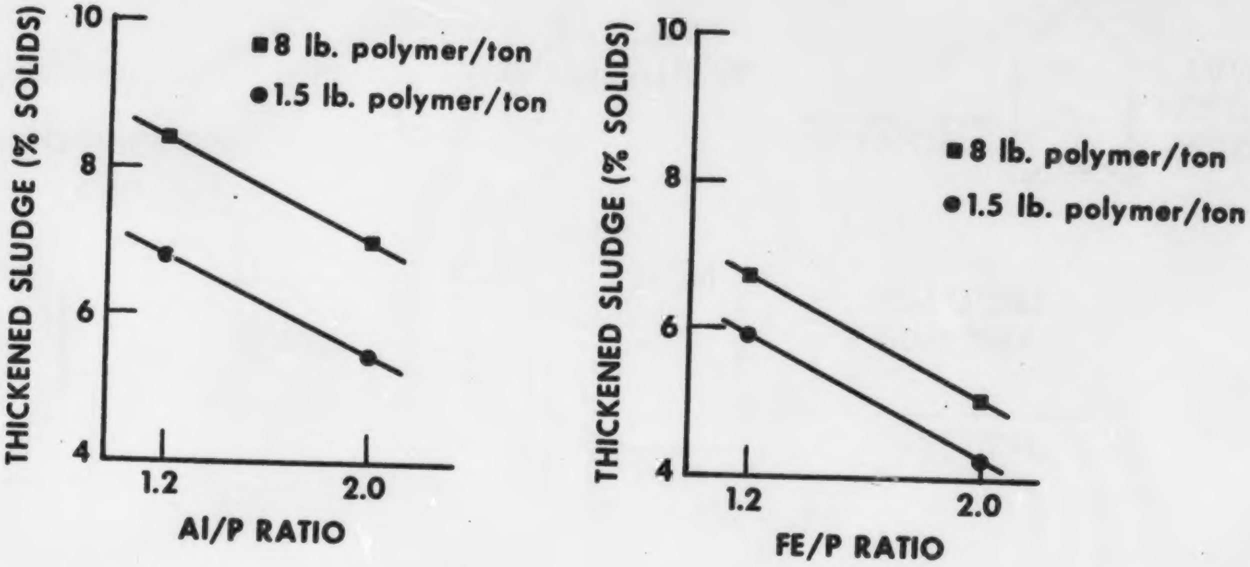

FIG. 6: EFFECT OF METAL/PHOSPORUS ATOMIC RATIO AND POLYMER DOSE ON THICKENED SLUDGE SOLIDS

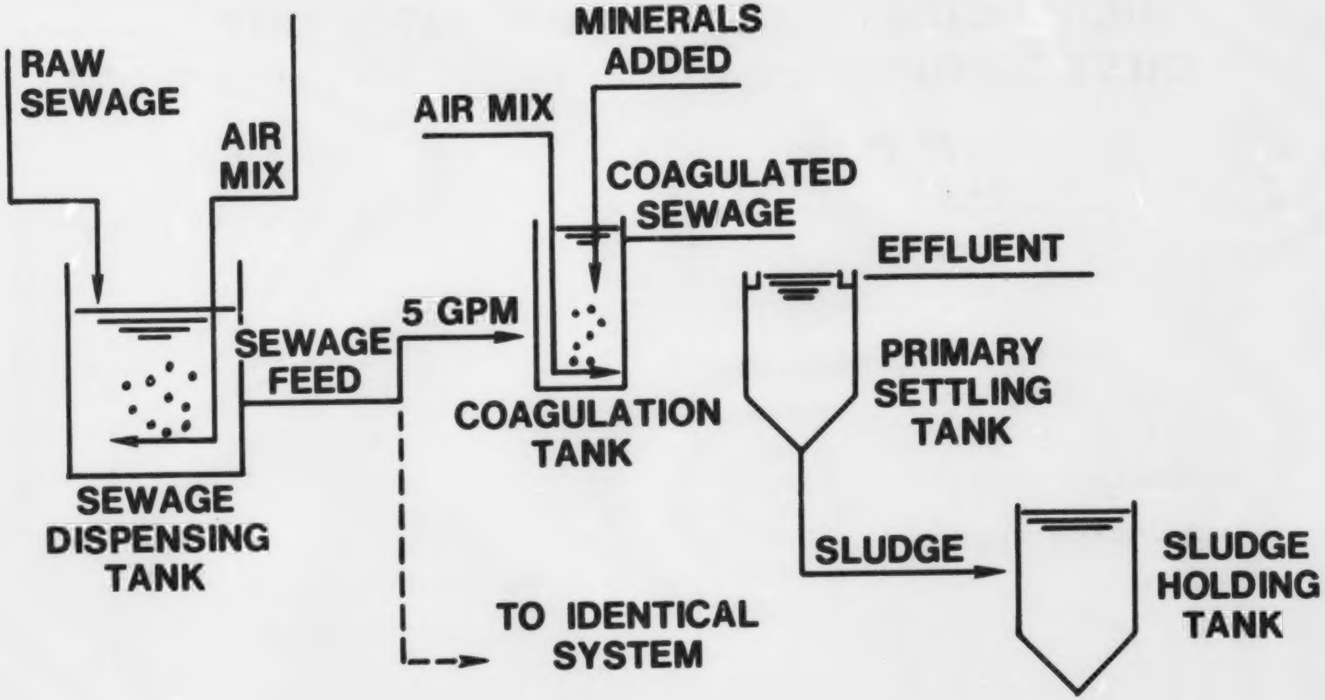
RAW SEWAGE
AIR MIX
SEWAGE DISPENSING TANK
SEWAGE FEED
5 GPM
TO IDENTICAL SYSTEM
AIR MIX
MINERALS ADDED
COAGULATED SEWAGE
COAGULATION TANK
EFFLUENT
PRIMARY SETTLING TANK
SLUDGE
SLUDGE HOLDING TANK

FIGURE 1
DUPLICATE PHYSICAL-CHEMICAL TREATMENT UNITS

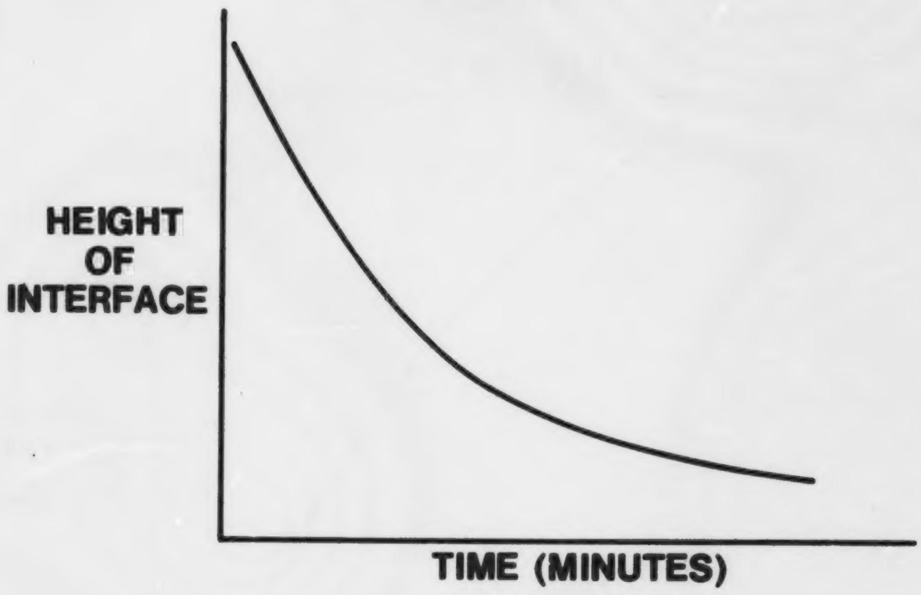

FIGURE 2
STANDARD SETTLING CURVE

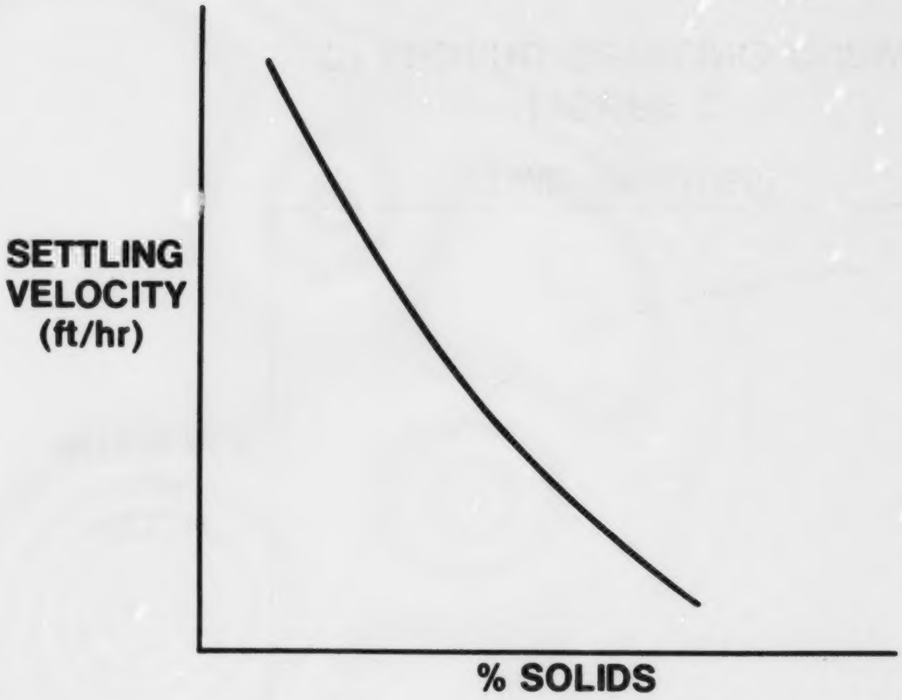

FIGURE 3
SETTLING VELOCITY

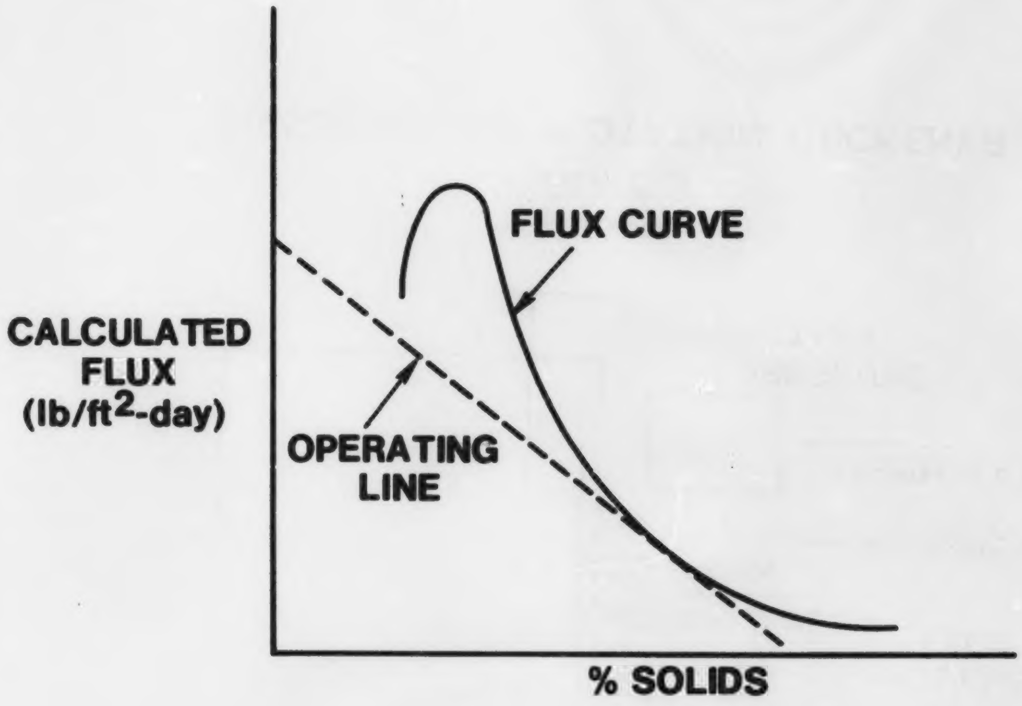

FIGURE 4
FLUX CURVE

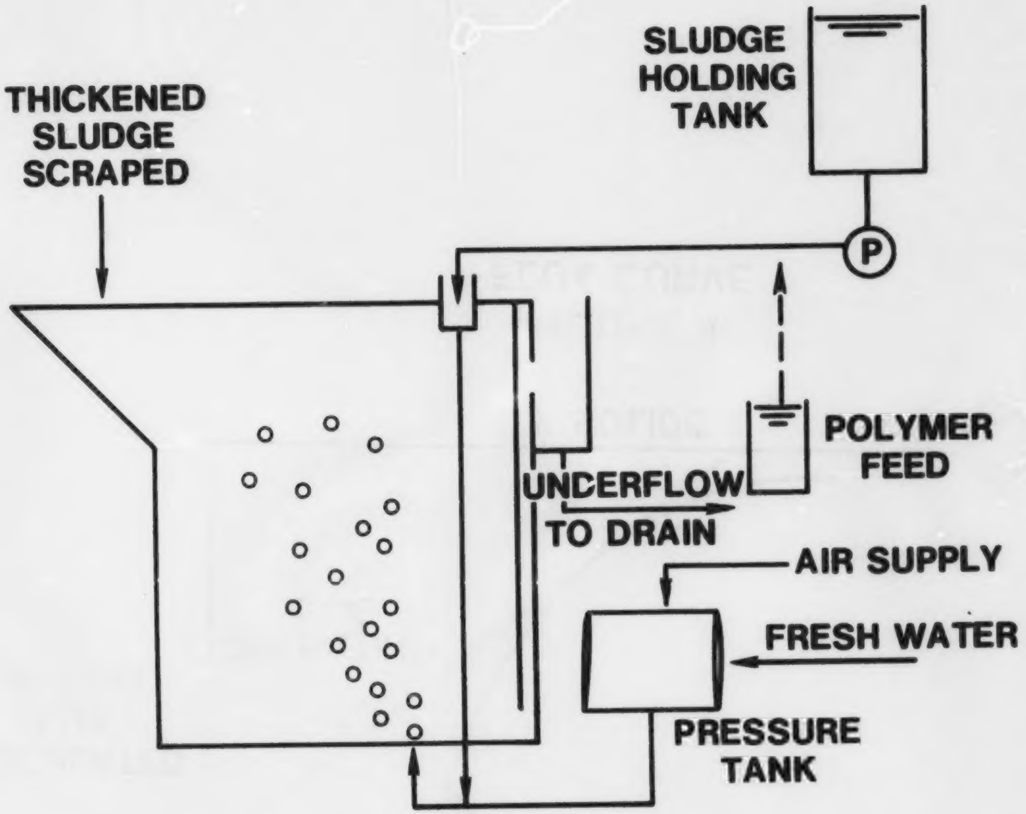
SLUDGE
HOLDING
TANK
THICKENED
SLUDGE
SCRAPED
P
POLYMER
FEED
UNDERFLOW
TO DRAIN
AIR SUPPLY
FRESH WATER
PRESSURE
TANK

FIGURE 5
DISSOLVED AIR FLOTATION THICKENER

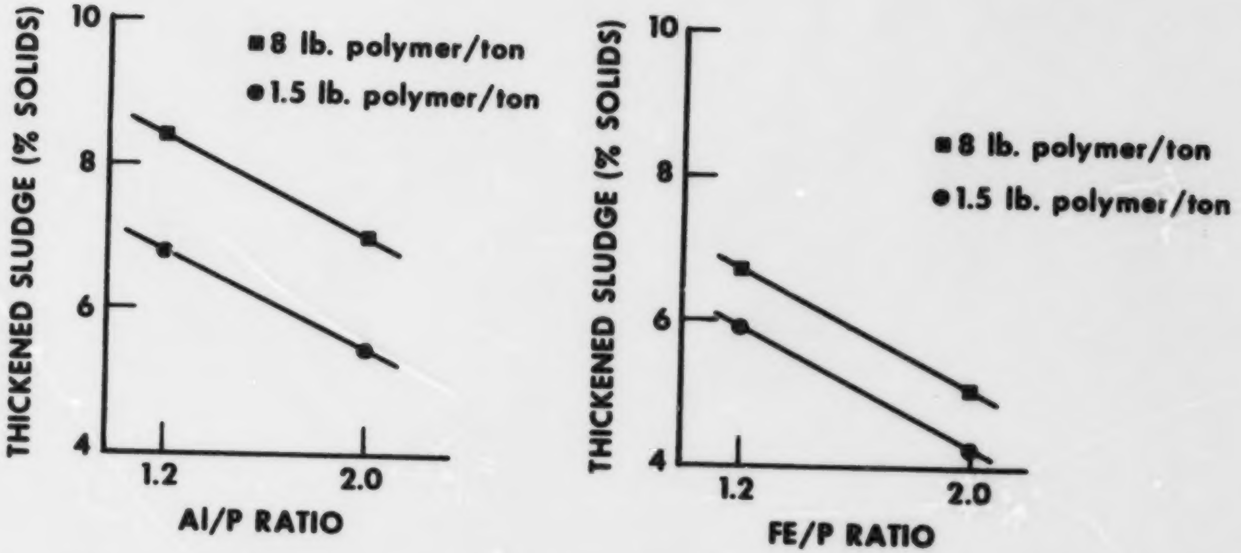

FIG. 6: EFFECT OF METAL/PHOSPORUS ATOMIC RATIO AND POLYMER DOSE ON THICKENED SLUDGE SOLIDS

SLUDGE INCINERATORS IN USE TODAY THAT MEET THE REQUIREMENTS OF STATE AND FEDERAL REGULATIONS*

Presented At

THE PRETREATMENT AND ULTIMATE DISPOSAL
OF WASTEWATER SOLIDS RESEARCH SYMPOSIUM

May I state here at the offset that the topic of my paper will not be as listed in your program. When first contacted with regard to this symposium, it was requested that I address my remarks to the different types of sludge incinerators in operation today that meet the requirements of State and Federal regulations and suggest proposals for the future. My paper has been prepared around this topic.

The fact that I do not profess to be a design engineer is perhaps reason enough not to address my remarks to the topic listed in the bulletin. My experience is that of supervision of construction. Trying to assure the client that the manufacturer's fabrication of a sludge incinerator will perform in accord with the designer's intent and will meet State and Federal regulations on such units. General remarks made herein are observations of field experience.

The actual destruction of sludge cake is not the real problem confronting us today. It is the conditioning of the sludge to place it in suitable form for presentation into the furnace and then the disposal of the residue of incineration--the ash and gases. While sludge conditioning is properly called a problem of incineration, it is separated in that it does take place prior to the presentation of the sludge into the furnace. On the other hand, the ash and gases formed as a direct result of the incineration do become part of the process of incineration.

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* R. L. Kaercher, Havens and Emerson Ltd.

Before going on, perhaps it would be well to identify State and Federal regulations. Proposed Federal rules on standards of performance as related to the Clean Air Act were issued in the Federal Register of June 11, 1973. The adopted rules were issued in the Federal Register of March 8, 1974. These rules, as one might expect, deal with stack emissions. The proposed rules of 1973 had as a basic criteria that stack emissions shall not contain particulate matter in excess of 0.031 grains per dry standard cubic foot and shall not exhibit 10 percent opacity or greater, except for two minutes in any one hour. Water vapor was recognized as not being a cause for failure of the regulations. In this regulation, a detailed test procedure was established for determining compliance with the regulations.

The State of New Jersey has classified sludge incinerators as Special Incinerators. As such, standards effective August 15, 1968, limit emission to not more than 0.1 grains of particles including ash per cubic foot of dry flue gas at standard conditions corrected to 12% carbon dioxide by volume excluding the contribution of auxiliary fuel. Smoke emissions limit has been established as to be not darker than No. 1 of the Ringelmann Smoke Chart. For new fires, such emission shall not be darker than No. 2 of the Ringelmann Chart for a period of three consecutive minutes.

With regard to ash, the State of New Jersey prohibits the emission of particles of unburned waste or ash which are individually large enough to be visible while suspended in the atmosphere.

The State does require a stack test, however, the method of performance is left to the descretion of the Department. Let me say here that any reference herein to the State of New Jersey is brought about by the fact that we are meeting today in this state. Regulations of any other state could be used in comparison.

There are six types of sludge incinerators in use in this country. The oldest and most popular is the Multiple Hearth. Flash Drying, Wet Oxidation, Fluidized Bed, and while perhaps not a pedigree sludge incinerator, the mixing of conditioned sludge cake with solid refuse and incinerating in a refuse incinerator. The last type on my list is the spray drying of sludge using waste heat from a refuse incinerator and burning the dried sludge in the incinerator.

The basic purpose of sludge incineration is to reduce the volume of sewage sludge to a minimum leaving a residue of inorganic, sterile ash. This must now be accomplished in conformance with ever increasing tighter regulations.

Disposal of the ash residue from any sludge incinerator is less of a problem than handling of the gases created. There are, however, problems to be concerned with. Hydraulic disposal of the ash to lagoons for settling is an accepted practice for handling of the ash from the multiple hearth, flash dryer or wet oxidation units. Ultimately, such lagoons must be excavated. Disposal sites must be available. Today this is perhaps not a problem but in years to come, this will be at least an economic problem as haul distances are increased. Consideration must also be given the characteristics of any ash cooling waters returned to the plant flow stream. This is especially important in the Fluidized Bed unit where all ash washing and degritting wash waters are returned to the plant flow stream. Inefficient capture of solids will put an extra solids loading on the plant system. Proper consideration must be given to the materials of construction used in the hydraulic piping system. Abrasive action of the ash requires special pumping equipment and special alloys in the fittings of the piping system.

Over the years, two types of emission controls have been developed, the dry type illustrated by cyclones, electrostatic precipatators and the bag filters. The wet type of impingement baffles and venturi scrubbers. Venturi scrubbers might be further defined as medium energy or high energy. High energy scrubbers usually having a pressure drop of 36 inches or more are more adaptable to refuse incinerators.

The wet scrubber has achieved greater success and efficiency and thus more acceptance than the dry. A review of stack emission tests performed prior to adoption of the new E.P.A. standards on modern sludge incinerators shows that today's standards can be met. Results of 28 separate stack tests from 21 existing sludge incinerators equipped with air pollution control facilities averaged particulate emissions of 0.027 gr/scf.

Emission tests reported on five plants studied in the E.P.A. report of "Sewage Sludge Incineration", Report No. EPA-R2-72-040, August, 1972 indicated that three of the five plants could meet the standards with existing equipment and the other two could be within the proposed limitations with relatively minor modifications.

It can be stated with assurance that properly designed sludge incinerators can meet the rigorous standards for particulate emissions.

Remember the bird cage on the stack to collect black birds --

Odor, while not mentioned above as part of Federal or State regulations, obviously must be controlled. The State of New Jersey stipulates that a special incinerator shall not be constructed or used which will result in odors being detectable by sense of smell in any area of human use or occupancy. Odor is controlled by subjecting gas flow to temperatures of 1300-1400° F. before emission from the units. This has not been a problem with any of the incinerator types mentioned here, except wet oxidation. The off gas from the separator is quite odorous with a high organic content. An after burner would definitely be required. The decant tank must be covered with proper treatment of off gases from such tanks provided. While this process provides more of a problem of odor control than others, it too can be handled.

We have all noted the clamor of the environmentalist over the past five or ten years who would have all the pollution ills of the country cleaned up overnight. Unfortunately, this is not possible. Usually the individual crying for such change has no feel for the problem as it really exists. Designs for sludge incinerators are not completed overnight. It is not unusual for an incinerator to be placed into actual operation three or four years after the initial authorization for design was given to the Engineer.

The Federal regulation stipulates that the standards of performance apply to sources, the construction of which was commenced after June 11, 1973. We have one plant for which the bids for construction were received on June 15, 1973. The design of this plant was accomplished much earlier under the regulation then in effect. Consideration should be given to installations such as this in evaluating the stack emission test results should they not meet the present requirements. The Federal regulations do at least stipulate that the standards will be applied to an established calendar date of construction start. One incinerator stack emission standard was changed by a regulatory agency after construction was started. This agency went further in that they did, in effect, dictate the type of equipment to be used for controlling stack emission. Even though the owner expended thousands of dollars to provide the equipment, the emission test failed to meet standards by a large margin. Fortunately, this is the only instance I can draw upon of such abuse of regulatory authority in this field. It does point up the dangers of individual interpretation of existing regulations.

Another procedure which must be given consideration in the construction of complete new wastewater treatment facilities, is the availability of proper sludge for incineration. Perhaps we have all experienced the wastewater treatment plant having been completed but not having the interceptor lines connected to provide flow or house connections be made so leisurely that flow was insufficient to produce adequate amounts of sludge to maintain incinerator operation for periods long enough to actually test at full load.

As each incinerator with its attendant emission control units is placed into operation, some new operating condition or design feature will be brought to the surface to be applied to the next unit. What will we have tomorrow--we are apparently experiencing a different sludge which will require new conditioning processes.

The impingement scrubber can work fine if proper alloys are used in the baffles, and velocities, temperatures and water quality are all controlled. Venturi scrubbers also do a proper job, again if it is recognized that the high velocity creates wear on the material. Proper amount of make-up water must be used to limit recirculation of ash laden cooling waters.

Reference has been made to the environmentalist who would clean up the air overnight. The cost for clean air should be closely examined before demanding compliance. High energy venturi scrubbers do perform very well. However, there is an accompanying cost of electrical power in using this unit due to the larger horsepower motors required. Is it worth the expenditure of electrical power to remove that additional grain from the stack emission?

Odor control has not proven to be a major problem. A few years back, the concensus of opinion seemed to favor the installation of after burners in the line to provide for temperature control of odors. On multiple hearth installations, it has been found that such burners are not required.

The presence of uncombined water or the white plume is recognized by many as not being a pollutant of the atmosphere, and indeed it is not. However, consideration must be given to this matter in evaluating the location of the sludge incinerator. Should the site be one subjected to conditions of atmospheric inversions, these plumes can raise havoc with traffic flow on adjacent highways. There is at least one plant site where an interstate highway was constructed within a few hundred feet of the existing plant. Under proper or perhaps improper atmospheric conditions, this highway is "fogged" out when the incinerator is operating.

It is perhaps too early to evaluate the effectiveness of the test methods established in the Federal Register, at least from my personal experience. Our office has only recently completed the first testing of an incinerator stack emission under these regulations and as yet, the results are not available. We do have four additional incinerators scheduled to be tested this year. Therefore, we do expect to have the proper experience to permit evaluation of the procedures on the basis of actual field installations in the very near future.

While six types of sludge incinerators and two basic types of scrubbers are available, experience has shown that the use of the multiple hearth furnace and a wet scrubber is providing the most practial answer to sludge incineration today. What will we have tomorrow? We are apparently experiencing a sludge having changing characteristics requiring new treatment processes in our wastewater plants. As we have increasingly restrictive regulations governing all actions, perhaps we need incinerator types not available today. It was a Buck Rogers fantasy to fly rockets only 40 years ago. Perhaps the new method will be to launch sludge ladened rockets to the sun's vicinity for complete destruction--odors and all.

It was mentioned a bit earlier that dangers do exist in the individual interpretation of regulations. Federal government control is divided into ten regions covering all states and lands under control of the United States Government. This, in effect, permits ten separate interpretations of the Federal regulations. There must be provisions made for some latitude in the enforcement of the rules. Otherwise, the same restrictive emission standards would be imposed for Guam as for New York City.

On the State level, somewhat the same condition does exist. Stack tests' methods are to be "as approved by the Department". Construction standards require new units to be of the multiple chamber type or of a type "approved by the Department". In this instance, a strict interpretation of the regulations would require the same incinerator design for Buena, in the openness of South Jersey, as would be required for Paterson. Most have observed the turnover of personnel in State, and where it is more in evidence, in Federal regulatory agencies. Replacement has not always been with personnel having the expertise necessary to properly review the submittal of an incinerator design.

For many years, the Engineer's designs have been restrained from most effective performance due to the lack of operators with the proper know-how or initiative to operate an incinerator. This problem is being overcome by operators and superintendents gaining on-the-job experience which they are now beginning to put to good use. Engineers have also gained design experience and further understanding of the problems of sludge incineration which is being passed onto the industry with each new unit being constructed. Is it not time, therefore, to research the problem of administration and enforcement of the standards? This would perhaps permit an intelligent understanding and agreement on what end results are really required and how fast they should be achieved.

Economic Considerations for Planning Sewage Sludge Disposal Systems*

by

Donn A. Derr
Victor Kasper, Jr.
Michael Gould
Emil J. Genetelli
**

Introduction

Municiaplities and other units of government have been devoting more and more of their scarce resources -- tax revenues and manpower -- to waste management. Much of this reallocation of tax revenue has been brought about in order to conform to Federal, state and local legislation attempting to maintain or increase the level of environmental quality.

Until recent years, decision-making bodies (private and public) have been able to avoid the total cost of waste disposal by passing part of the cost onto a second party through the use of common property goods like water and air. This is more commonly known as an externality or market failure. That is, the user of a resource or product does not pay for its "total" cost. In many instances, it is very difficult to compute "total" cost because of non-monetary costs which lack adequate units of measurement for lost social value.

Also, many of these wastes are becoming a competitive source of raw materials as they become more plentiful and virgin materials more scarce and expensive. The wastes have essentially become "misplaced resources." Through recycling and other means of recovery, the cost of waste management and the reliance on virgin materials is reduced.

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*Presented at the Symposium on Pretreatment and Ultimate Disposal of Wastewater Solids, Rutgers University, New Brunswick, N. J., May 21-22, 1974.

**Associate Professor and Research Associate, respectively, Department of Agricultural Economics and Marketing; and Graduate Student and Professor, respectively, Department of Environmental Science, Cook College, Rutgers University, New Brunswick, N. J.

This paper suggests procedures for the estimation and evaluation of costs for alternative (waste) sewage sludge disposal systems. First, the basic components or modes of these systems are identified, then the flow of activity, the components of cost, and, finally, the interpretation of results.

Basic Components

The basic components or modes for sewage sludge disposal are identified as: (1) transport, (2) dewatering, (3) storage, and (4) site disposal. Within each of these four basic components are several possible options or alternatives. For example, the transport mode can include: (1) tank truck, (2) dump truck, (3) pipeline, (4) barge, and (5) rail.

In the various systems discussed in this paper, a dewater/no dewater option is incorporated. This option is provided at one of two locations: the point of origin (treatment plant), and just before disposal. Dewatering can reduce transport costs and storage facilities; and for certain disposal methods, it is necessary for proper operation.

The storage option provides flexibility for varying weather conditions, equipment breakdown and work distribution.

Basic disposal methods include: (1) incineration, (2) landfill, (3) ocean disposal, and (4) land disposal. For purposes of discussion, sludge solids concentration will be confined to two levels -- 5 and 30 percent.

Transportation Mode

The transport component can be of three possible types: (1) land modes, (2) water modes, or (3) a combination. For land transport: (1)

tank truck, (2) dump truck, (3) rail, and (4) pipeline are possible. The selected method will depend in large part on the distance from treatment plant to disposal site, the concentration of solids, and the annual volume of sewage sludge. For example, a tank truck would be needed for 5 percent solids and a dump truck for 30 percent solids.

The truck transport generally has a lower initial investment cost but will have higher operating costs relative to rail or pipeline transport for most levels of design volume. A rail transport system involves the construction of a rail spur and car fees. Pipeline requires a heavy investment in right-of-way, construction, and restoration (landscaping) costs.

Water transport of sewage sludge can be by either barge or pipeline outfall. Again, costs for barge and pipeline are analogous to rail and land pipeline costs, respectively.

Dewatering Mode

A dewatering option is provided for technical and economic reasons. By dewatering, additional equipment can be considered, like dump trucks, and incineration. Also, since large quantities of water are removed, the volume of sludge is reduced, and, subsequently, costs.

Three methods are in current use today: (1) sand beds, (2) centrifuge, and (3) vacuum filters. Sand beds have low initial costs but operating costs are high because of maintenance such as the removal of dried sludge and operating difficulties caused by heavy rainfall and freezing conditions.(1)

Centrifuges are affected much less by varying weather conditions and have lower operating costs. However, the initial investment will generally be higher. Centrifuges are plagued by the fines that do not settle out well when returned to the treatment system. This problem can be overcome with greater bowl speed, retention time, and chemical treatment. (2) Vacuum filters generally operate with less trouble, particularly with chemical-treated sludges. (3)

Storage Mode

Storage of sewage sludge can occur primarily at the sewage plant or the disposal site, or both. Storage permits flexibility in terms of equipment usage, work loads, and weather variability. It also reduces environmental impact. With storage capacity, sludge can be disposed at a rate in keeping with the rate of soil capacity.

Disposal Methods

Disposal methods include: (1) incineration, (2) land disposal, (3) landfill, and (4) ocean disposal. Each have advantages and disadvantages. For example, incineration requires dewatering, air pollution control devices, residue disposal, and relatively little land use. Land disposal obviously requires extensive land area but permits utilization of the nutrients contained in the sludge and may be less of a problem to the ground water than landfill.

Disposal at a landfill site after considering all disposal costs is probably cheaper than land disposal. However, there is the problem of leachates, possible contamination of ground water, aesthetics and the future availability of landfill sites.

Ocean disposal costs will be affected by the location of the sewage treatment plant. Two transport modes may be required -- land and water. Also, the effect on marine life and recreational facilities, as well as varying legislative policies makes this method somewhat uncertain.

It is also possible to use more than one disposal mode in order to minimize costs and/or the impact on the environment. For example, it may be possible to use ocean outfall to certain limits and then apply the balance through land disposal with nutrient recovery. This can be handled through an optimization technique (cost minimization) similar to linear programming. This will be discussed in a later section.

Flow of Activity

The flow of sludge or the combined modes of activity of various alternative combinations are portrayed in Figures 1 through 4. Figure 1 illustrates the combined activities for incineration, Figure 2 for land disposal, Figure 3 for landfill, and Figure 4 for ocean disposal.

The flow charts are divided into three segments with each segment representing modes of activity at a given location. The three segments include: (1) the origin (sewage treatment plant site), (2) the transport mode, and (3) destination (disposal site). This three-way grouping is made to indicate the possible location of modes. For example, in Figure 1 (incineration), dewatering plus storage facilities could be located at the sewage treatment plant. Also, storage and dewatering could be located at the disposal or incineration site, depending upon the availability of land area. The origin and destination are then connected by the transport mode. The location of dewatering and storage facilities can, and probably will, affect costs because sludge would be transported at 30 percent solids if dewatered at the sewage plant, and at 5 percent solids if dewatered at the incineration site. More equipment and time would be required for 5 percent solids.

The flow of activity in Figure 2 indicates that sludge can be disposed with or without dewatering by land disposal. The equipment used for the disposal activity, however, will depend on whether the dewatering option

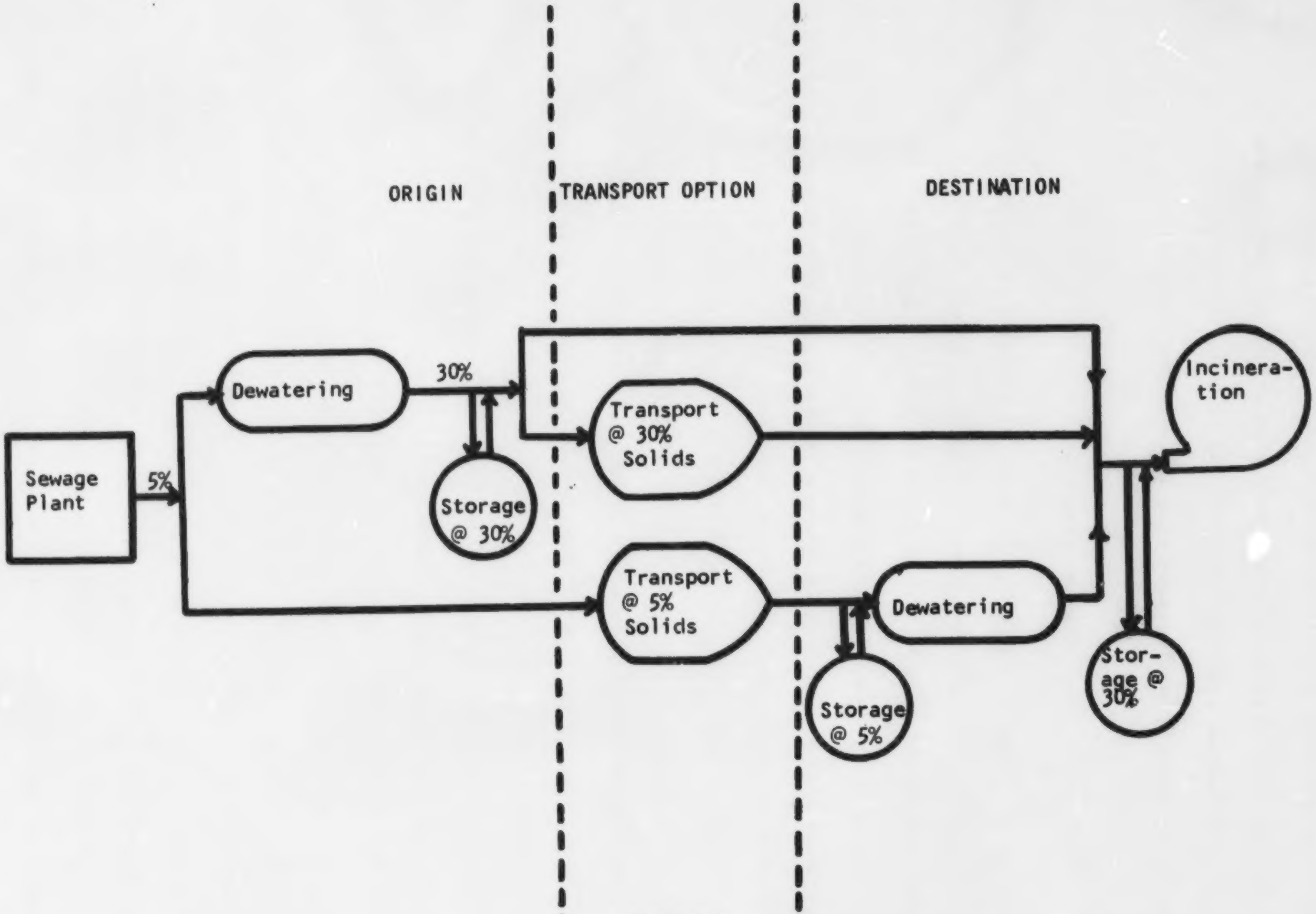

ORIGIN
TRANSPORT OPTION
DESTINATION
Sewage Plant
5%
Dewatering
30%
Storage @ 30%
Transport @ 30% Solids
Transport @ 5% Solids
Storage @ 5%
Dewatering
Incineration
Storage @ 30%

FIGURE 1

FLOW CHART OF ACTIVITY FOR INCINERATION DISPOSAL OF SLUDGE

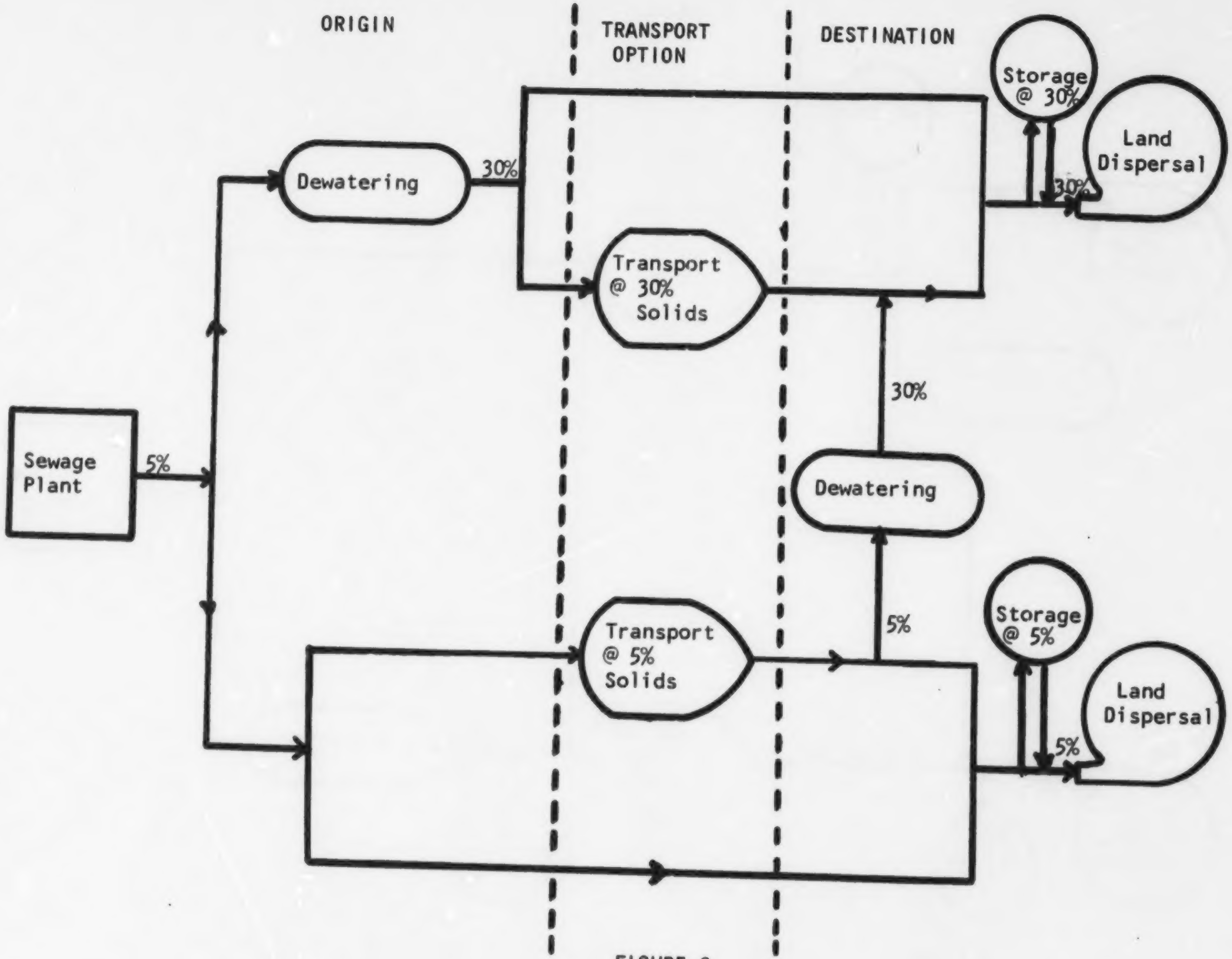

ORIGIN
TRANSPORT OPTION
DESTINATION
Sewage Plant
5%
Dewatering
30%
Transport @ 30% Solids
Transport @ 5% Solids
30%
Dewatering
5%
Storage @ 30%
Land Dispersal
30%
Storage @ 5%
Land Dispersal
5%

FIGURE 2
FLOW CHART OF ACTIVITY FOR DISPOSAL OF SEWAGE SLUDGE BY LAND DISPERSAL

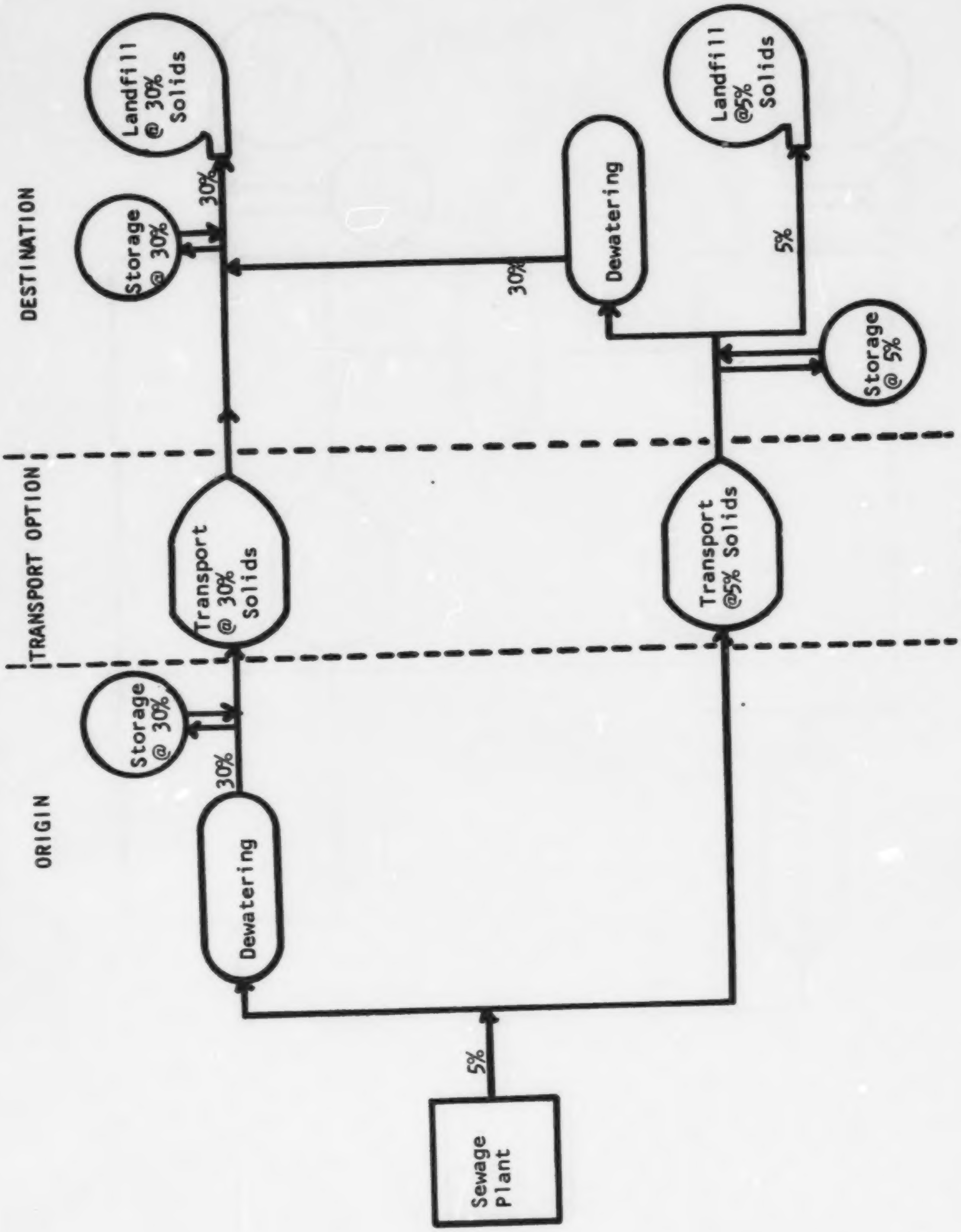

ORIGIN
TRANSPORT OPTION
DESTINATION
Sewage Plant
5%
Dewatering
30%
Storage @ 30%
Transport @ 30% Solids
Storage @ 30%
30%
Landfill @ 30% Solids
Transport @5% Solids
Storage @ 5%
Dewatering
30%
5%
Landfill @5% Solids

FIGURE 3

FLOW CHART OF ACTIVITY FOR DISPOSAL OF SEWAGE SLUDGE BY LANDFILL

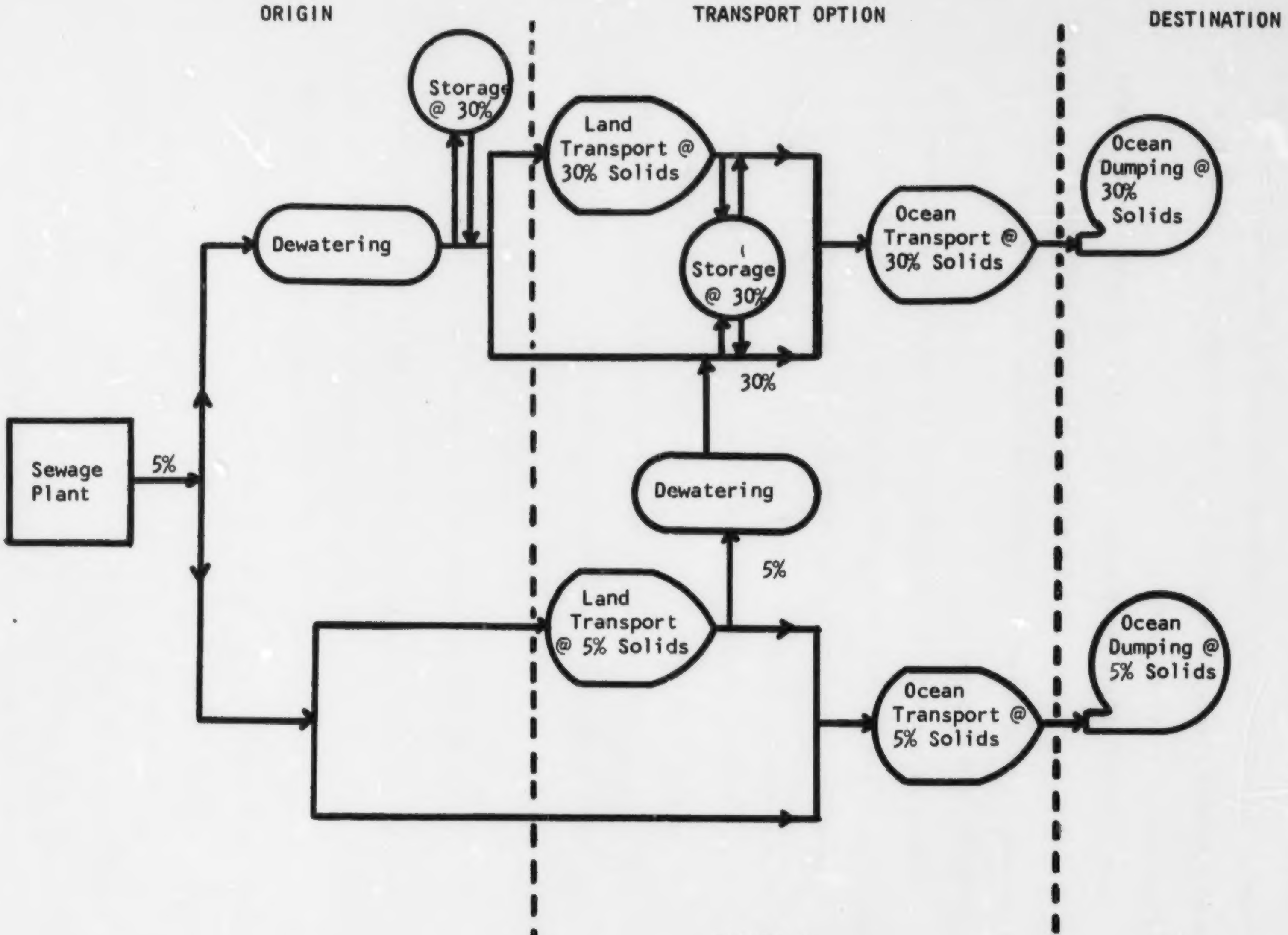

ORIGIN
TRANSPORT OPTION
DESTINATION
Sewage Plant
5%
Dewatering
Storage @ 30%
Land Transport @ 30% Solids
Storage @ 30%
30%
Ocean Transport @ 30% Solids
Ocean Dumping @ 30% Solids
Dewatering
5%
Land Transport @ 5% Solids
Ocean Transport @ 5% Solids
Ocean Dumping @ 5% Solids

FIGURE 4

FLOW CHART OF ACTIVITY FOR DISPOSAL OF SEWAGE SLUDGE BY OCEAN DUMPING was selected. Here, again, dewatering can occur either prior to or following transport. In addition, if the disposal destination is adjacent to the sewage treatment plant, one alternative is for sludge to be loaded directly into the land disposal equipment.

Land disposal offers the greatest opportunity for recycling of sludge into the soil using a crop like Bermuda grass for nutrient removal.

Disposal through landfill is quite similar to the activities discussed under land disposal.

The flow of activity for ocean disposal is presented in Figure 4. Ocean disposal can be used to dispose dewatered or nondewatered sludge. Two forms of transportation may be necessary. If the sewage plant is located adjacent to a water transport facility, only an ocean transport capability will be required. On the other hand, if the sewage treatment plant is located a distance from the coast, land transport is required in addition to the ocean transport. Dewatering can take place either before or after land transport. Also, ocean transport can be implemented by barge and by pipeline (outfall).

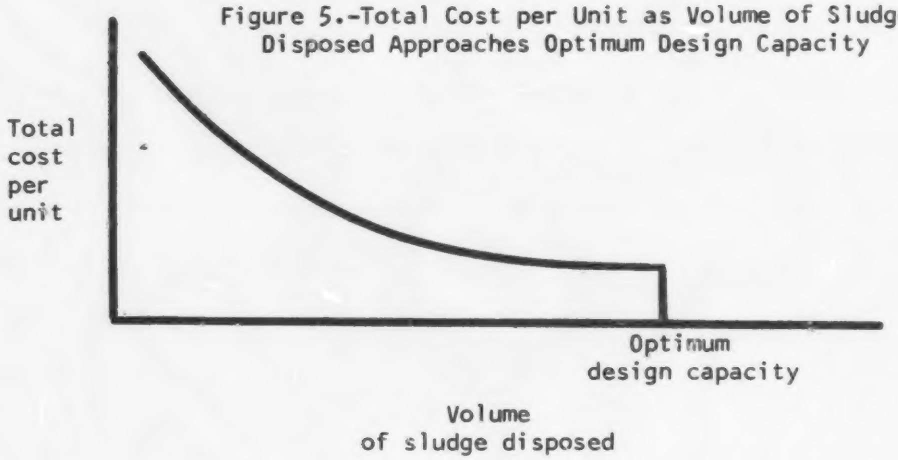
Estimation and Components of Sewage Sludge Costs

In most cases of computer modelling and resource allocation, the objective function is to maximize profit or minimize costs. Here, the basic objective is to minimize the cost of sewage sludge disposal, taking into account environmental constraints which may be dictated by law or public welfare considerations.

Costs can be placed into three categories: (1) capital investment, (2) operating or variable costs, and (3) fixed costs. Capital investments are those costs or outlays required before the first unit of sewage can be disposed. Included are such costs as engineering and planning fees, buildings, equipment, land, cost of initial financing and certain personnel costs, like administration.

Variable costs include hourly labor, electricity, chemicals, petroleum products, and repairs and servicing. These costs vary directly with the quantity of product being processed. Fixed costs include those expenditures that are incurred independent of the volume of processing, like insurance, reserves for replacement, administration, interest and taxes. Initial capital investment in machinery, buildings and structures prorated over its physical life and reflected in the reserves for replacement is much of fixed cost. In essence, initial capital investment cost indicates the amount of funds required to start an operation. The design capacity associated with the initial capital investment results in a specific annual cost made up of fixed and variable costs. On a per unit basis, fixed costs will vary depending upon how close to optimum capacity the system is operated while per unit variable costs will remain relatively constant. Figure 5 indicates the typical relationship of total cost per unit as the system is operated closer to optimum design capacity. The total cost per unit drops rapidly at first and then approaches a minimum when it reaches design capacity.

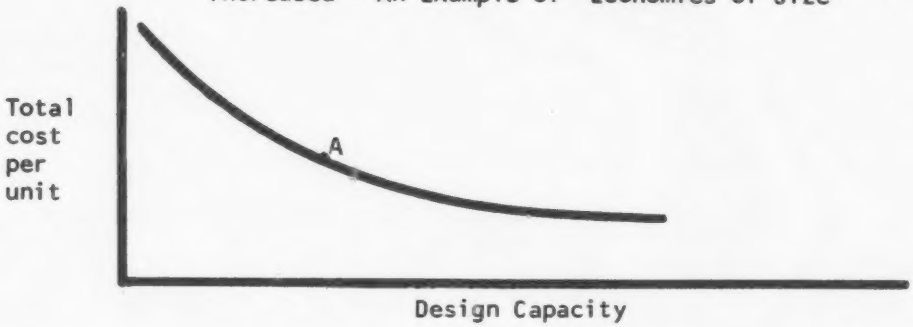
Figure 5.-Total Cost per Unit as Volume of Sludge Disposed Approaches Optimum Design Capacity



A relationship similar to that presented in Figure 5 may exist for total unit cost and design capacity. In many processes, the incremental capital investment per unit declines as design capacity is increased. In other words, the overall investment per unit of capacity decreases for systems with larger design capacities. This has been referred to as "economies of size". This concept should not be confused with the relationship between total cost per unit and sludge disposal operating at less than design capacity as described by Figure 5.

Figure 6 is a representation of the relationship between total cost per unit and design capacity illustrating the concept of economies of size. Point A represents a system such as that presented in Figure 5 operating at optimum design capacity.

Figure 6.-Total Cost per Unit as Design Capacity is Increased - An Example of "Economies of Size"



Evaluation

To evaluate the alternative methods of sludge disposal, in addition to investment and cost data, consideration must be given to budgetary constraints, environmental impact and risk associated with each alternative. Which method a community can select is limited by the funding available and its potential revenue over time. The selection can also be affected by legislation restricting the environmental impact. Finally, the potential risk (probability of making a wrong decision) associated with changes in factor-price relationships, technology, growth rate, revenue, environmental legislation, and loading capacity of the soil, must also be considered.

1. Budgetary Constraints

After estimating the required investment, and, subsequently, costs, a decision-making unit must determine the amount of present and future resources which must be allocated to sludge disposal. The investment and cost estimates can then be compared to available resources. This cash-flow analysis would eliminate alternatives outside the decision-making unit's budgetary constraint. Table 1 indicates a general cash flow situation over the operating life of a particular system. The cash outflow of each year is compared to the available resources allocated to sludge disposal. The cash outflows of alternative systems designed for the community's needs could then be compared to the resources available for allocation to sludge disposal. The cash outflow of alternative systems which were equal to or less than the available resources would lie within the planning unit's budgetary constraint.

It must be noted that 0_t (Table 1) represents net cash outflow for each period, t. Revenue has been subtracted from gross cash outflow. This revenue could result from the sale of a possible product produced from the disposal activity. As a varient of the land disposal technique, sludge could be heated, dried, and sold as a soil conditioner. This method has been tried and has met with limited success. The main obstacle is market development costs. Another possibility of off-setting sludge disposal costs is the recovery of heat from incineration for plant heating. In any case, potential revenue from possible sales should be considered a cost offset. As the price of fertilizer increases the sale of sludge as

Table 1.-Example of a Cash-Flow Comparison of a Sludge Disposal System Given Resource Constraint

Cash Outflow O_t	Expected Resources Allocated to Sludge Disposal R_t
O_1	R_1
O_2	R_2
.	.
.	.
.	.
O_T	R_T

R_t = f (tax revenue, credit, state, and Federal aid)

T = operating life of the system

O_t = variable costs + fixed costs - depreciation + equipment replacement outlays - revenue from sales.

a soil conditioner may increase in importance as a source of revenue for off-setting sludge disposal costs.

Alternatives within the planning unit's budgetary constraint could then be evaluated with respect to the present worth of the cash outflow over time. The alternatives with a smaller present worth of total costs could then be further evaluated. The following formula could be used for this purpose:

$$P_W = I + \sum_{t=1}^{T} \frac{C_t}{(1+r)^t}$$

where:

P_W = present worth of total cost of the system

I = initial capital investment

C_t = variable cost + fixed cost - revenue depreciation - interest charge

r = discount rate

T = life of disposal system

The discount rate can be estimated by the rate of return the planning unit could have received by investing each C_t in alternative investments. The discounted costs in time period t, $\frac{C_t}{(1+r)^t}$, is interpreted as the present worth of a payment, C_t, made in period t. The higher the discount rate, the lower the value of these future operating costs. Thus, a capital intensive alternative which has a large initial capital investment but relatively low operating costs will more likely be selected when the discount rate is high. On the other hand, a labor intensive alternative with a lower initial capital investment and a higher operating cost will more likely be selected when the discount rate is low.

The present worth of the discounted cash outflows, $\sum \frac{C_t}{(1+r)^t}$, can be interpreted as the amount of capital which would have been invested at the going rate of interest so that the periodic expenditures are met.

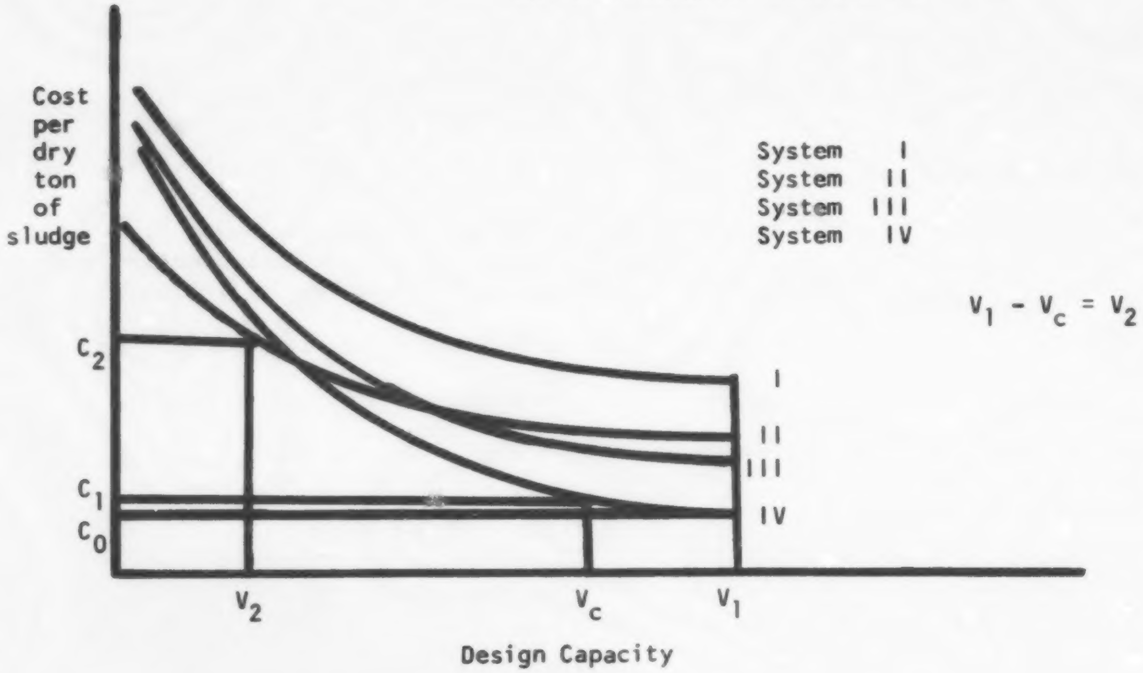
2. Environmental Impact

The impact of the disposal method on the environment is difficult to measure. Assumptions must be made in order to estimate how environmental effects will influence the selection of a disposal method or methods. Environmental constraints can be placed in the planner's selection process in two ways: (1) the planner can be given restrictions on how the environment can be affected from members of the political entity for which they are planning, and (2) legislation may set limits in terms of rates of disposal, available disposal sites, or types of equipment which can be used. For the following discussion, it is assumed that a given level or constraint is provided.

Figure 7 illustrates the problem of selecting a sludge disposal system under an environmental constraint. A planning unit must provide for the disposal of V_1 volume of sludge. Under no constraints it would select the system which had the least cost at a design capacity of V_1. In the hypothetical example illustrated by Figure 7, the no constraint selection would be system IV. This system provides a disposal capacity of V_1 at a cost per unit of C_0.

Now consider the case where an environmental constraint is placed on the planning unit by legislation. The hypothetical law limits the volume which can be disposed by system IV to V_c. This volume, V_c, becomes an environmental constraint.

Figure 7.-Hypothetical Cost per Dry Ton for Four Alternative Sludge Disposal Systems Illustrating Different Levels of Economies of Size and Environmental Constraints



Given this constraint, the question arises as to what is the next best alternative. A number of alternatives are possible. One alternative is to select system III which has the next lowest cost per unit at design volume V_1. A second alternative would be to select a two-system approach such as disposing of volume V_c by system IV and volume $V_1 - V_c$ by system II, where system II has the least cost at a design capacity at volume $V_1 - V_c$. There is a large number of multi-system approaches which can be considered. The question is which approach has the least total cost per unit for a design capacity V_1. A method which can be used to determine the least cost alternative(s) is the recursive programming approach. The environmental constraints and cost functions as illustrated in Figure 7 would be inputs into the program. The program is then used to search both individual and multi-system alternatives within these constraints and cost functions to arrive at a solution which approximates the least cost alternative.

The question of what should be the environmental constraints is a question yet to be determined. Once determined, however, the constraints can be evaluated in terms of their effects on decision-making by the procedure suggested above.

3. Risk

The element of risk can be an important factor in evaluating the feasibility of alternative methods. Relative factor prices, technology, legislation, and available resources may change. All of these can affect the criteria for selection of sludge disposal methods.

To take these factors into consideration, the importance and probability of each source of uncertainty could be estimated. The minimum cost alternatives could be estimated for a variety of factor-price relationships, technology, legislation, and resource constraints. This is a form of sensitivity analysis. The factors which produced radically different costs for alternatives could then be evaluated in terms of their probability of occurrence. The minimum cost alternative could then be selected using the expected value of these sensitive factors.

The procedure of estimating the probability of possible changes would be a key factor in determining the minimum cost alternative. These estimates would rely heavily on the judgement of the planners, historical information and life span of the project.

The life span of the system is particularly important. Systems with longer operating lives are more risky in the sense that factors are more likely to change as the length of time considered increases. In evaluating two systems, the expected value of a sensitive factor for a long-lived system could be different from a system which was relatively short-lived.

Another risk which could be considered is that of the use of inaccurate estimates of population growth over time. Inaccuracies in these estimates could result in not considering systems which would otherwise have been selected as least cost alternatives. The weighted expected value approach could be used under these circumstances.

A simplified example of how these factors could be incorporated is illustrated in Table 2.

The decision-making unit has determined the probability of wage rates, W_i, to be, P_i, during the operating life of a particular system. The expected value of wages is then ΣP_iW_i. This value provides a means of estimating the cost of a disposal system while taking into consideration the possible instability of a factor to which per unit cost is particularly sensitive. Other factors can be evaluated in a similar manner.

Table 2.-Determination of the Expected Value of Wages for a Hypothesized Disposal System

Wage rate	Probability of each wage	
W_1	P_1	P_1W_1
W_2	P_2	P_2W_2
W_3	P_3	P_3W_3
Expected wage		ΣP_iW_i

where:

$P_1 + P_2 + P_3 = 1$

P_i = probability of W_i $\quad i = 1, 2, 3$

Hypothesized Solutions

Given the components or activities described in this paper, estimates of costs were obtained from past literature, and previous research of the authors, and are presented in Table 3. These are estimates of total costs. The conditions under which each cost was estimated differed in such areas as design volume, distance transported, soil absorption capacity, etc. However, they provide the basis for a hypothesized solution to the selection of a sludge disposal system.

The least expensive solution based on Table 3 would be for a sewage treatment plant located at barge-docking facilities to barge sludge at 5% solids for a distance less than 30 miles and dispose the sludge in the ocean. This solution does not provide for storage facilities.

The most expensive alternative is for sludge to be transported over land by tank truck for 17 miles or more, stored at 5 percent solids concentration, and disposed by land disposal.

Table 3.-Typical Estimated Total Costs per Dry Ton for Component Activities, 1974

Methods	Cost per Dry Ton[1/] ($)
1. Dewatering	
A. Vacuum filter	31.00
B. Centrifuge	26.00
C. Sand beds	30.00
2. Land Transport 5%	
A. Tank truck	3.02/mile
B. Railroad	.25/mile
C. Pipeline	1.55/mile
3. Land Transport 30 %	
A. Dump truck	.65/mile
B. Railroad	.25/mile
4. Ocean Transport 5%	
A. Barge	.20/mile
B. Outfall	.60/mile
5. Ocean Transport 30%	
A. Barge	.03/mile
6. Storage	
A. 30%	2.30
B. 5%	14.00
7. Disposal 5%	
A. Ocean disposal	--
B. Landfill	3.00
C. Land disposal	20.00
8. Disposal 30%	
A. Ocean disposal	--
B. Landfill	3.00
C. Land disposal	10.00
D. Incineration	12.00

[1/] Cost has been adjusted for inflation at 4% per year.

Source: Burd, R.S., A Study in Sludge Handling and Disposal, U.S. Department of the Interior, Publ. pp. 179-514, Washington, D.C., May, 1968.

(cont. next page)

Thus, the cost of alternatives presented by Table 3 ranges between $6 per dry ton for barging and $85 per dry ton for land disposal.

Given the costs in Table 3, however, all that can be determined is that current costs of sludge disposal can range between these values. Little or no comparison of systems can be made without additional information. The information required for further evaluation is the total investment, the variable and fixed costs for the relevant range of design capacities, the budgetary constraints of the decision-making unit, the institutional and/or environmental constraints, and the possibility of change in unstable factors for which costs are particularly sensitive. Once this information has been estimated, the array of alternative systems can be adequately

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Source: (Table 3, cont.)

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Kasper, Victor Jr., Michael I. Gould, Donn A. Derr, and Emil J. Genetelli, "Procedure for Estimating the Cost and Investment Required for Sludge Recycling through Land Disposal", Unpublished Report of Department of Agricultural Economics and Marketing and Department of Environmental Sciences, Rutgers University, New Brunswick, New Jersey, August, 1973.

considered by the decision-making unit for selection of the least cost alternative necessary to meet a region's sludge disposal needs.

Alternative sludge disposal systems must be evaluated under comparable design conditions and with consideration of budgetary constraints on environmental impact and risk. To make decisions on the partial information such as that in Table 3, increases the possibility that sub-optional or less efficient solutions will be selected.

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(2) White, W.F., and J.E. Burnes, "Continuous Centrifugal Treatment of Sewage Sludge", Water and Sewage Works, Volume 109, No. 10, October, 1962, pp. 384-386.

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Future Problems in Sludge Production and Handling Systems

Joseph V. Hunter

The salts of iron (ic) or aluminum, or calcium hydroxide (lime) can be employed in waste water treatment in two major capacities. Firstly, they can be employed as coagulants for the removal of particulate materials, and as such can be used for secondary effluent upgrading or on waste water itself as part of a complete physio-chemical treatment process.

Secondly, they can be employed as precipitants (or adsorbents) in phosphate removal processes, and as such may also be employed for concurrent particulate matter removal. In addition, alum can also be added directly to activated sludge tanks to achieve phosphate removal. Wherever employed, the inorganic compounds formed by their reactions will be found in the sludge, and it is the purpose of this paper to indicate what modifications and problems in present sludge treatment and handling processes might be expected from their employment.

Sludge Composition

As the composition of the sludges will depend on the nature of the coagulant or precipitant, its dosage, the extent of phospnate removal and the waste water composition, particularly the concentration and value of the particulates, generalizations on the composition of such sludges are difficult to make. However, a knowledge of these factors and of the nature of the reactions involved does enable certain predictions as to what would be the expected composition of the sludge.

1. Alum Precipitation:

Alum (aluminum sulfate hydrate) can be used to precipitate phosphorous as follows: $Al_2(SO_4) + 2\ Na_3PO_4 = 2\ AlPO_4 \downarrow + 3Na_2SO_4$. At the same time, alum will be reacting with the waste water alkalinity as follows:

$$Al_2(SO_4)_3 + Ca(HCO_3)_2 = 2Al(OH)_3 \downarrow + 3CaSO_4 + 6CO_2\ .$$

In theory, one mole of aluminum should precipitate one mole of phosphorous, or, on a weight basis, 0.87 parts of aluminum should precipitate one part of phosphorous. As alum contains about 9% aluminum, 9.7 parts of alum would be required to precipitate one part of phosphorous or 3.1 parts of phosphate (1). Due to the reaction with alkalinity and possible reactions with the organic constituents, the actual dosage required to precipitate one part of phosphorous or 3.1 parts of phosphate would be about 2 parts of aluminum or 22.2 parts of alum. Dosages, therefore, in the range of 22.2 mg/l alum per 3.1 mg/l phosphate will give about a 95% phosphate reduction.

If the suspended solids removed, the % volatiles in the suspended solids, the phosphate concentration, and the alum dosage are known, it should be possible to predict the sludge composition. For example, an average dosage of 87 mg/l alum was employed to coagulate a trickling filter effluent containing 48 mg/l suspended solids. The alum dosage reported could precipitate

3.9 mg/l of phosphorous, or form 15 mg/l of aluminum phosphate. If the aluminum required to react with the phosphate is subtracted from the alum dosage, the remainder represents the amount of aluminum that precipitates as the hydroxide. If the trickling filter effluent solids are 70% volatile then the following composition would be predicted:

particulate organics	43 mg/l
particulate inorganics	5 mg/l
aluminum phosphate	15 mg/l
aluminum hydroxide	12 mg/l

If, on ignition, the aluminum hydroxide forms aluminum oxide (Al_2O_3), then the volatile solids of this sludge would be 63%, which agrees with the reported value of 60% (2). In this case it was not necessary to know the phosphate concentration, as it was unquestionably above 12 mg/l.

2. Iron Precipitation

The reactions of iron salts in water are quite similar to those of alum. If ferric chloride is used as a precipitant, then the phosphorous precipitation reaction would be an analogous calculation procedure which can be employed to determine sludge composition. In a study on waste water composition, 70 mg/l (anhydrous) ferric sulfate was used to coagulate Highland Park, N.J. domestic wastewater (4). The volatile suspended solids were 125 mg/l and the nonvolatile solids 21 mg/l. This dosage of iron would form 10.7 mg/l $FePO_4$, and the remainder would form 41.5 mg/l $Fe(OH)_3$. Assuming complete suspended solids precipitation, and that on ignition the $Fe(OH)_3$ would form Fe_2O_3, the calculated volatile solids of a dry sludge would be 68%. This again agrees quite well with the observed value of 70%

Chemical coagulation, however, can do more than simply remove suspended solids. In the domestic waste water study noted above, about one-fourth of the soluble COD was removed along with the particulates, and detailed sludge analysis indicates that the additional organics removed by chemical coagulation (i.e., over that of physical separation) were not analogous to the usual particulate constituents, confirming to some extent their different mechanisms.

3. Lime Precipitation

Due to the many reactions involved, lime precipitation is a far more complex process than alum or ferric chloride precipitation. In addition to the effluent organic and inorganic particulates, the precipitation reactions will add the following constituents:

a. Hydroxyapatite

This is the phosphorous removal precipitate, and is formed as follows:

$$5Ca(OH)_2 + HPO_4^{=} = Ca_5OH(PO_4)_3\downarrow + 3H_2O + 6OH^-$$

b. Calcium Carbonate

Calcium carbonate is generated in two ways through lime addition. The first is through reaction with the alkalinity (the lime softening reaction)

$$Ca(OH)_2 + Ca(HCO_3)_2 = 2CaCO_3\downarrow + 2H_2O.$$

The second is through reaction with carbon dioxide

$$Ca(OH)_2 + CO_2 = CaCO_3\downarrow + H_2O.$$

c. Magnesium Hydroxide

If the pH is sufficiently high (i.e., $\simeq$11), the magnesium in solution will also be precipitated as follows:

$$MgSO_4 + Ca(OH)_2 = M_2(OH)_2 \downarrow + CaSO_4.$$

In addition to these materials we may also find insoluble impurities in the lime itself.

Unlike precipitation with iron and aluminum salts, phosphate reduction employing calcium hydroxide is mainly a function of the pH to which the system is raised by the lime addition. Thus, the dosage of lime required is more a function of the wastewater alkalinity than the phosphate concentration. Thus, if the pH of the system is raised to about 11, then phosphate reductions of over 98% can be achieved (3).

Despite the complexity of the sludge formed it has been possible to predict the sludge composition (5). This was done on the basis of wastewater suspended solids removed, the phosphorous removed as $Ca_5OH(PO_4)_3$, that all magnesium was precipitated as $Mg(OH)_2$ and that calcium carbonate precipitation could be calculated by employing a mass balance around the calcium ion as follows (as equivalents):

$$CaCO_3 \text{ pptated} = Ca(\text{in}) + Ca\ (\text{dose}) - Ca\ (\text{out}) - Ca\ (\text{apatite}).$$

On this basis, the following results were obtained:

Constituent	Observed %	Calculated %
Organic	28.9	24.4
Hydroxy apatite	5.9	4.8
Magnesium hydroxide	6.7	3.9
Calcium carbonate	52.1	60.6
Balance	6.4	6.0

Considering the complexity of the system, these results show reasonably good agreement, and thus lime sludge composition can also be predicted with reasonable accuracy.

Sludge Production

Using the same type of computations employed to calculate the sludge composition, it should be also possible to predict the dry weight of sludge produced. The Highland Park, N.J. data mentioned in the previous section represented results on a daily average basis (flow composited over an extended period. The sludge produced was 198 mg/l, which is equivalent to 1.7 lbs/1000 gallons. This was, however, for a phosphate removal of 4 mg/l phosphate-P. As the wastewater probably contained about 8 mg/l phosphate-P, the sludge concentration for complete phosphate removal would have been about 250 mg/l (doubling the hydroxyapatite value of 10.7 and the aluminum hydroxide value of 41.5) equivalent to 2.1 lbs./1000 gallons. As the average flow over this period was 1.7 mgd, then the daily sludge production for about 50% phosphate reduction would have been 2890 lbs., and for essentially complete phosphate reduction it would have been about 3570 lbs.

These values are not out of line with those to be expected of such sludges ( ). For example, alum processes tend to produce 2-2.5 lbs. of dry solids/1000 gals. of wastewater treated, ferric chloride processes 2.5-3 lbs./1000 gals., and lime processes 6-13 lbs./1000 gals.

Sludge Water Holding Capacities

Although the dry weight quantities of sludges produced by alum, ferric chloride or lime coagulation or precipitation may be predicted with some degree of reliability, the water holding capacities of the sludges have essentially been determined by empirical study rather than from basic considerations. In general, alum and iron sludges have the greatest water holding capacity, and lime sludges least (6). Typical ranges are:

	Sludge Type			
	Activated	Alum	Ferric Chloride	Lime
Thickened Solids, %	2-4	2-4	3-4	10-15
Dewatered Solids, %	15-20	12-18	12-20	25-50

Although lime sludges thicken and dewater better than alum or ferric chloride sludges, more lime sludge is produced on a dry solids basis Thus we have a somewhat self-compensating effect, and the result is that the wet volumes of sludges produced may not be too different (6), and the following results might be expected:

	Type of Sludge		
	Alum	Ferric Chloride	Lime
lbs. wet sludge per 1000 gallons wastewater treated	11-21	12-25	12-26

Needless to say, such generalizations are always hazardous. Local conditions, the nature of the wastewater or plant effluents treated, the purpose of the chemical treatment all influence the values noted, but they do represent typical results if such can be said to exist.

Sludge Digestion

Although chemical precipitation or coagulation unquestionably increases the dry weight of sludge, the dewatering characteristics of the sludges are such that there are not particularly large increases in the volumes of sludges produced. For example, there have been reports that the use of alum (7) or lime (8) did not increase the volumes of sludge produced in the treatment process. In such cases, additional digestor capacity would not be required from a strictly volumetric loading standpoint. However, if larger sludge volumes are produced, this would decrease the digestor retention time and thus could adversely influence digestion unless digestor capacity was increased.

It is also possible that the presence of the reaction products of the chemical precipitants or coagulants could interfere in the digestion process itself. Although aluminum precipitated phosphate has been reported to give no interference in sludge digestion (7, 9), the addition of 1549 mg/l aluminum (as the hydroxide) to a sludge containing 1.8% solids gave a 15% decrease in gas production over a 37 day period (10). It was concluded that as little as 100 mg/l aluminum (as the hydroxide) could adversely affect sludge digestion.

Lime coagulation sludge also indicated an adverse effect on digestion processes (5). Using a lime sludge containing 50-55% inorganic constituents, it was observed that both volatile matter destruction and gas production were adversely affected. For example, after 15 days, volatile matter reductions in the lime sludges were about 35%, while volatile matter reductions in the control sludge were about 50%. The volume of methane produced per part of volatile matter destroyed were essentially the same; but the gas produced by the lime sludge digestion contained considerably less carbon dioxide, due to its interaction with the alkaline sludge constituents. Whether this was due to a limitation on the rate or the extent of digestion is not clear, but the authors suggest that it may be rate considerations. If so, this could possibly be compensated for through increased retention time. Even with relatively poor digestion, the sludge was not too objectionable due to its high alkalinity (eg., low free hydrogen sulfide levels, etc.). However, as there are problems of stratification due to the denser calcium carbonates, and as the lime sludge itself is relatively stable and can be disposed of advantageously employing other technologies, the digestion of such high inorganic contents lime sludges is questionable.

As a considerable part of the interest in the use of alum, ferric chloride or lime lies in their use as phosphate precipitants, one of the major concerns must be whether or not sludge digestion causes phosphate resolubilization. Studies have indicated no phosphate release from the digestion of alum precipitated sludges (11); nor is this surprising as there is no obvious resolubilization mechanism. However, it was also reported that no solubilization of phosphate occured when iron precipitated sludges were digested (11). This is somewhat surprising as phosphate recycling in stratified lakes has long been explained by the reduction of ferric phosphate to a more soluble ferrous phosphate under anerobic conditions.

Lime precipiation is carried out at pH values above 11, but during digestion the production of carbon dioxide and volatile acids will reduce this to about 7.7 (5). Theoretically, this should release substantial quantities of the precipitated ortho phosphate, but the digestion study previously described indicated that only about 1.5% of the phosphate was resolubilized. However, the reduction of pH did solubilize essentially all the magnesium hydroxide in the lime precipitated sludge.

Precipitant or Coagulant Regeneration

Whenever chemically coagulated or precipitated sludges are discussed, the question of coagulant or precipitant regeneration inevitably arises. This is due to the cost of these materials, the problems they do or may cause during treatment and disposal, as well as the greater quantities of sludges produced for disposal.

At present, iron is not in extensive use and substantial work has not been done in studying its regeneration, although processes may be applicable Lime and alum regeneration, however, have been extensively studied, and the discussion will be limited to these two.

1. Lime Regeneration

Lime sludge contains coagulated or precipitated organic materials, calcium carbonate, calcium hydroxyphosphate (hydroxyapatite), magnesium hydroxide, and other insoluble inorganic constituents (silica, clays, etc.). Regeneration procedures usually involve dewatering to 50-55% solids then recalcifying at 1800°F. what occurs here is the loss of organics by combustion, the conversion of calcium carbonate to calcium oxide, and the conversion of magnesium hydroxide to magnesium oxide. The miscellaneous inorganics and the hydroxy apatite remain relatively unaffected.

This mixture can then be rehydrated and agin used as a precipitant. However, it does contain both inorganics and hydroxy apatite. As the lime is recycled, these could build up and cause problems. Although calcium hydroxide is considerably more soluble than the other constituents, the lime is fed as a slurrey and thus this property of calcium hydroxide is not usually employed at present.

Culp and Culp (3) have described a possible way out of this difficulty. The calcium carbonate sludge is denser than the phosphate sludge, and they propose a series of two centrifuges; the first to remove the carbonate (and some organic and magnesium hydroxide). The second to remove the phosphate sludge and the remainder of the organics and magnesium hydroxide. The first is recalcined and returned to the process as a lime slurry. The second is incinerated and then processed for disposal. However, there still are reservations about the build up of inert, insoluble inorganics in return hydrated recalcined lime sludges (11).

2. Alum Sludges

Aluminum sludges are simpler, containing only sluminum hydroxide, aluminum phosphate, and the wastewater or effluent particulates removed during coagulation or precipitation. As aluminum is amphoteric, two regeneration processes can be employed:

a. Alkaline regeneration

At a pH of 11.9, the precipitated aluminum hydroxide will dissolve to form soluble aluminate as follows:

$Al(OH)_3 + NaOH = NaAlO_2 + 3H_2O$.

At the same pH, aluminum phosphate will also dissolve as follows:

$AlPO_4 + 4NaOH = NaAlO_2 + 2H_2O + Na_3PO_4$.

Thus the phosphate will go into solution as well as the aluminum. Fortunately, the phosphate can be precipitated with calcium chloride as follows:

$2Na_3PO_4 + 3CaCl_2 = Ca_3(PO_4)_2 + 6NaCl$.

At pH values above 12, about 90-95% of the aluminum can be recovered, and be substantially freed of its phosphate contents. As many organic materials are anionic and are insoluble in neutral or acid pH ranges but soluble in alkaline pH ranges (eg., stearic acid), alkaline regeneration will dissolve or solubilize some of the sludge organics. The remainder of the sludge particulates remain in the alkaline suspension which in turn must be treated and disposed.

Alkaline regeneration thus produces the highly alkaline sodium aluminate. The optimum pH for aluminum precipitation/coagulation is below 7 however, and thus it is not too surprising that sodium aluminate has been reported to be a poor coagulant or precipitant (3). It has also been noted that such regeneration is more costly than the original alum, but this must be weighed against the lesser sludge produced for disposal.

b. Acid regeneration

When the pH of an alum sludge is reduced to 2.5-3, approximately all the aluminum will go into solution as follows:

$$2Al(OH)_3 + 3H_2SO_4 = Al_2(SO_4)_3 + 6H_2O$$

$$2AlPO_4 + 3H_2SO_4 + Al_2(SO_4)_3 + 2H_3PO_4$$

Unfortunately, in an acid media it is not possible to remove the phosphate by precipiation with calcium chloride. Thus, acid regeneration of alum, although it produces an acceptable precipitant or coagulant and is economically advantageous (less costly than alum) (3), cannot be employed for phosphate removal systems. It may be possible to remove the solubilized phosphate by ion exchange (3), but this considerably lessens the economic advantages of the acid regeneration process, and must result in ion exchange regenerants for disposal as well. As in the previous regeneration system, inorganic and organic particulates will remain in a acidic sludge for dispoal or treatment, but in this case little solubilization of organics is to expected.

SUMMARY

The use of alum, ferric chloride or lime as coagulants or precipitants unquestionably increases the total dry weight of the sludges produced. Due to the better dewatering characteristics of certain of these sludges (i.e., lime sludge), the volumes of the sludges produced may not be larger than those produced before the introduction of the coagulation or precipitation processes.

These sludges contain higher inorganic contents, and their composition is fairly predictable from a knowledge of the reactions involved and the amounts of particulates and phosphate removed. The presence of these precipitants or coagulants may adversely affect anaerobic digestion, but as there are many other aludge treatment and disposal methods applicable, this may not be too significant a disadvantage.

Both lime sludges and alum sludge can be reprocessed to recover the precipitants. Lime carbonate sludges may be segregated from the phosphate sludges, recalcined hydrated, and returned to the process. Alum sludge may be recovered using acid or alkaline processes, but the acid process while economically feasible and producing a satisfactory coagulant, cannot be used for phosphate removal systems. Alkaline regeneration can be employed for phosphate removal systems but seems to result in a poorer coagulant.

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EPA'S POSITION ON OCEAN DISPOSAL IN THE NEW YORK BIGHT

Richard T. Dewling, Richard D. Spear, Peter W. Anderson and Randy J. Braun

Ocean disposal in the New York Bight is a significant environmental issue, particularly when one realizes that on a national basis more than 70 percent of all municipal sludge dumping occurs in the New York Bight. Similarly, industrial waste dumping represents more than 60 percent of the national problem.

EPA's responsibility for managing and monitoring ocean disposal activities only began in April 1973, the effective date of the Marine Protection, Research, and Sanctuaries Act. Our actions, however, considering this short time period, have been significant in the New York Bight and have resulted in reducing the environmental impact associated with this practice. Highlights of our actions and progress during the past 15 months are as follows:

....Eight (8) industrial applications/permits have been denied.

....Forty-seven (47) previous dumpers have been forced to choose other environmentally acceptable alternatives, thus phasing out ocean disposal. An additional twelve (12) industries will be phased out by June 1975.

....Industrial wastes, previously dumped at the 12-mile sewage sludge site, are now all being transported off of the continental shelf and disposed of at the 106-mile site.

....Digester cleanout from sewage treatment plants, which contains a high percentage of floatables, must now be disposed of off the continental shelf. Previously, this material was dumped twelve (12) miles from our shore.

....No new industrial or municipal dumpers, other than those using this method of ultimate disposal prior to the passage of the Ocean Dumping Bill, have been approved for using any of the sites in the New York Bight.

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....All dumpers are now required to provide EPA with a detailed chemical and biological analysis of the waste materials being discharged into the ocean. No such requirement existed prior to April 1973.

....With the cooperation of the U. S. Coast Guard, which is responsible for "police type" monitoring of vessels using the disposal sites, EPA has initiated a vigorous enforcement program. Violations of permit conditions, including "failure to notify", premature dumping, and non-segregation of municipal and industrial wastes, have been discovered and legal action initiated.

....Municipalities in the metropolitan area have been notified of our intention to move the present sewage sludge dumping ground in 1976, or earlier, if our monitoring programs indicate an environmental threat. EPA's long-range goal is to completely phase out ocean disposal of municipal sludge by 1981.

Present Problems —— Real or Imaginary??

The furor created in the press has undoubtedly caused concern among the citizens of the metropolitan area, and naturally, with the regulatory agencies, namely, the state, county and local health departments, and the United States Environmental Protection Agency. Information presented in the media, in our opinion, lacks the technical support needed to draw the types of conclusions that have been reported —— namely, that there is a massive uprooting and movement of the sludge mass from the present 12-mile disposal site to the shore of Long Island.

Rather than try to attack the technical validity of the statements made by others, we would prefer to have you examine the problem as a whole, thus hopefully recognizing the complexities of resolving this problem in an environmentally acceptable manner.

At the present time, 5.6 million cubic yards per year of municipal sludge (see Table I) are being disposed of within the New York Bight. Ocean disposal of municipal sewage sludge in the New York Bight was initiated by the Passaic Valley Sewerage Commissioners. The dumping of this material at sea was permitted by the United States Army Corps of Engineers under the Captain of the Port proviso of the Rivers and Harbors Act of 1888. By 1977, the date by which municipalities are required to provide secondary treatment, in accordance with

TABLE I

OCEAN DUMPING IN THE NEW YORK BIGHT

70% OF ALL MUNICIPAL SLUDGE DUMPING IN U. S.

1973 - 5,590,144 cu. yds.
60% from NYC
7% Nassau & Westchester
33% New Jersey

60% OF ALL INDUSTRIAL WASTES DUMPED IN U. S.

	Volume (yd^3)	Volume (gal)
Allied Chemical	68,128	13,795,992
American Cyanamid	151,371	30,652,640
Chevron	40,677	8,237,184
E. I. duPont	291,528	59,034,482
Hess Oil	8,959	1,814,172
Modern Trans. Co.	99,537	20,156,250
NL Industries	3,135,947	635,029,200
1973 TOTAL	3,796,147	768,719,920

DREDGE SPOIL FY'73 - 11,818,250 cu. yds.

the mandates in the 1972 Amendments to the Federal Water Pollution Control Act, it is anticipated that the sludge volume generated will increase significantly. By 1980, it is expected to reach 18 million cubic yards per year. How do we handle this large volume of sludge in an environmentally acceptable manner? The alternatives available to us are as follows:

(1) Disposal in a regional incinerator or incinerators, located at a site where air quality impact would be minimal. Offshore sites merit serious consideration. Power generation and the burning of other solid wastes along with sludge should be considered in order to enhance the economic feasibility of this type project.

(2) Controlled disposal to the marine environment. Other sites and disposal techniques are presently being investigated by EPA and NOAA. More than $7,000,000 of Federal funds are being expended to study the acute environmental stresses associated with this practice, as well as determining the long-term effects which includes measuring the impact on marine organisms and the food web.

(3) Disposal at remote landfill sites. If this alternative is considered feasible, pretreatment techniques such as sludge dewatering and leachate treatment will be required.

Let's dwell on this alternative for a moment. It is our opinion that the technical solution to resolving this complex environmental problem of ocean disposal is here today, i.e., sludge could be pumped or shipped to the strip mines in Pennsylvania or to open land areas in New York State. One of the problems with this alternative, assuming that the environmental consequences associated with this type of operation could be resolved —— and they can —— is overriding the political and social constraints associated with the various governmental agencies. In other words, the people in Pennsylvania, or for that matter upstate New York, do not want the waste products from people outside of their geographical, social, or political boundaries. Thus, one is forced to resolve his environmental problems in his own "back yard". It is interesting to note that the City of Chicago has just won the 1974 American Civil Engineering Award for its project involving the transportation of sludge to a land site 200 miles south of the City. Chicago purchased 11,000 acres and reportedly has satisfactorily solved their sludge disposal problem in this way.

(4) Disposal as a soil conditioner or fertilizer. Several years ago, EPA, recognizing that this was a viable alternative, provided $200,000 to Ocean County, New Jersey, to demonstrate the feasibility of land recycling on the sandy soils of the Pine Barrens. The results of this study will be available by June of next year.

Pre-treatment —— Prerequisite to Disposal

It must be recognized that for sewage sludge to be disposed of by any of the above-mentioned alternatives, it must be properly pretreated to meet EPA requirements for heavy metals and toxic components, as mandated in the 1972 FWPCA Amendments. All land disposal alternatives must consider the potential pollution problems of groundwaters in the disposal areas. In addition to requiring sophisticated techniques to meet stringent air quality standards, sludge incineration will still produce an ash —— 0.35 million cubic yards per year by 1980 —— that must be handled. Thus, without further study of these alternatives, it is impossible to predict whether any of these approaches, as applied to resolving the problem in this metropolitan area, are more or less damaging to the environment than the present practice of ocean disposal. It is for this reason, therefore, that EPA funded a two year, $200,000 study with the Interstate Sanitation Commission to look at these and other alternatives and to recommend, by June 1976, the soundest and most environmentally acceptable alternative to resolving this problem.

Phase Out Ocean Disposal by 1981

Our approach to assessing and managing the present practice in the New York Bight has been at two levels —— short-term and long-term. With regard to the latter, we have developed and implemented a plan —— with an ultimate goal of phasing out ocean disposal by 1981 —— which is technically feasible, environmentally sound, and one that, hopefully, has not been influenced by media-induced hysteria and emotionalism. Figure I highlights this plan.

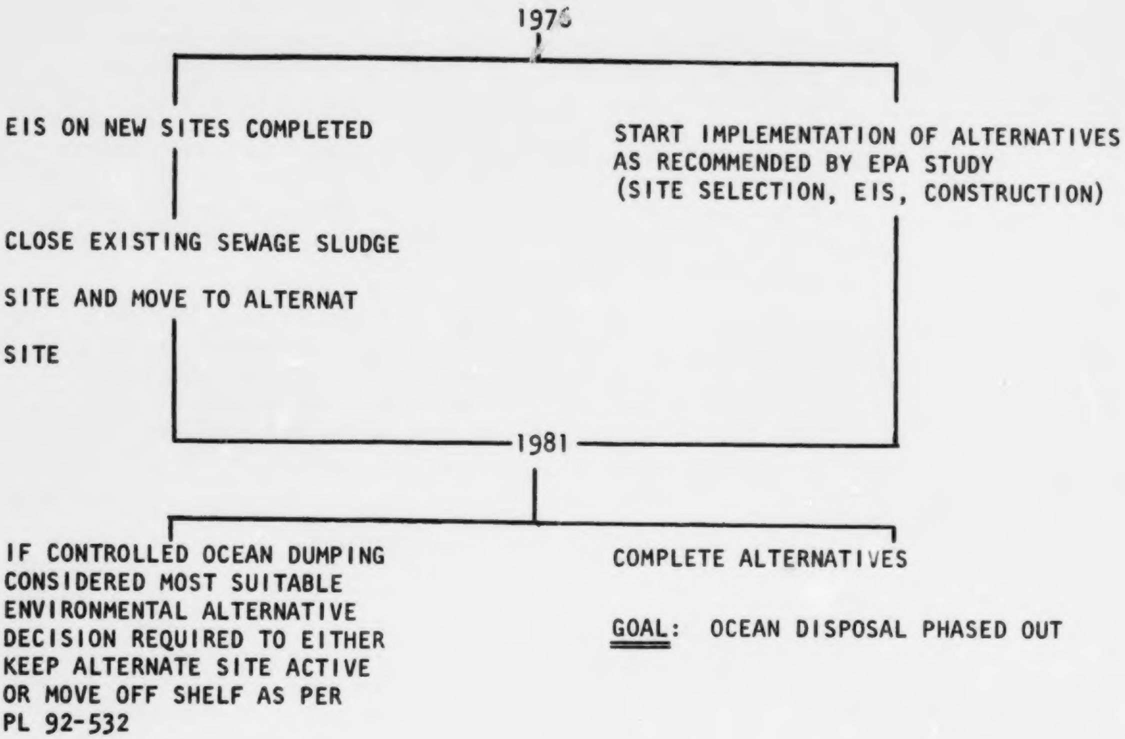
FIGURE I

LONG RANGE PLAN FOR PHASE OUT

	1970	N. Y. BIGHT STUDY - U. S. CORPS OF ENGINEERS: SANDY HOOK LABORATORY
	1971	EPA CONDITIONS CONSTRUCTION GRANTS
JUNE	1972	EPA OCEAN COUNTY STUDY: $200,000, 3 YRS; LAND RECYCLING
APRIL	1973	OCEAN DUMPING BILL EFFECTIVE - EPA PERMIT PROGRAM STARTED
DECEMBER	1973	EPA SEGREGATES WASTES AT DUMP SITES
APRIL	1974	EIGHT (8) INDUSTRIAL PERMITS/APPLICATIONS DENIED OR WITHDRAWN: FORTY-SEVEN (47) INDUSTRIAL DUMPERS PHASED OUT: TWELVE (12) INDUSTRIAL DUMPERS REQUIRED TO PHASE OUT BY 6/75
APRIL 10,	1974	MUNICIPALITIES NOTIFIED OF MOVING TO ALTERNATE SITE BY 1976
JUNE	1974	NOAA INITIATES FIELD STUDIES OF ALTERNATE SITES FOR EIS
JUNE	1974	EPA INITIATES "ENVIRONMENTALLY ACCEPTABLE ALTERNATIVE" STUDY, $200,000, 2 YRS
	1974	NOTIFY CORPS OF ENGINEERS OF SITE CHANGE FOR "POLLUTED" DREDGE SPOIL
JULY	1974	EPA AWARDS $540,000 TO RAYTHEON FOR ADDITIONAL EIS STUDIES

FIGURE I (Continued)

LONG RANGE PLAN FOR PHASE OUT



1976

EIS ON NEW SITES COMPLETED

CLOSE EXISTING SEWAGE SLUDGE

SITE AND MOVE TO ALTERNAT

SITE

START IMPLEMENTATION OF ALTERNATIVES
AS RECOMMENDED BY EPA STUDY
(SITE SELECTION, EIS, CONSTRUCTION)

1981

IF CONTROLLED OCEAN DUMPING
CONSIDERED MOST SUITABLE
ENVIRONMENTAL ALTERNATIVE
DECISION REQUIRED TO EITHER
KEEP ALTERNATE SITE ACTIVE
OR MOVE OFF SHELF AS PER
PL 92-532

COMPLETE ALTERNATIVES

GOAL: OCEAN DISPOSAL PHASED OUT

The short-range solution is somewhat more complex, not from the technical standpoint, but because it is heavily impacted by emotional reactions to what has been reported in the press. In that regard, our studies, which are presented in detail in Appendix A, can be summarized as follows:

(1) Renewed claims and reports by private scientists that the sludge mass has uprooted itself and is moving toward the Long Island shore at an alarming and unprecedented rate have not been substantiated. In fact, our surveys show contradictory data, not from the standpoint of disclaiming the findings of private researchers —— since EPA has also found "black mayonnaise" one-half mile from shore at random locations —— but in technically disagreeing with the conclusions drawn by these individuals.

(2) Studies on and along the beaches of Long Island, New York and New Jersey clearly indicate that these waters, based on Federal and State bacteriological standards, are safe for contact recreation. The absence of pathogens —— disease-causing bacteria —— in these waters provides further verification of the excellent quality.

(3) The "black mayonnaise" found at random locations one-half mile from the beach has been identified as natural decaying organic material.

(4) Results of cruises, perpendicular to, and through the dumping grounds, suggest that the problems reported in the press are related to inland occurrences and not associated with a massive movement of material from the disposal site. If, in fact, there was a massive movement towards shore, one would expect to find a gradual diminution of pollution levels from the disposal site to the shore area. (See Appendix A)

(5) The leading edge of the sludge mass, associated with the sewage sludge disposal site, is still located approximately 5-1/2 to 6 miles from the shore of Long Island, thus negating the urgency to move the present disposal site before the scheduled 1976 closure.

(6) A recommendation to immediately move the present sewage sludge site, without simultaneously moving the dredge spoil site, which presently influences and impacts the sludge site, would be shortsighted and environmentally unwise. To simply move one without the other would not solve any problems, except satisfy those who personally feel that the sludge dumping ground is the major cause of all environmental ills in the New York Bight.

(7) A recommendation to immediately move the present disposal sites —— sewage sludge and dredge spoil —— founded on emotionalism rather than technical data, could prove environmentally disastrous.

EPA is pledged to protecting the environment. That is our role as professionals, citizens, and as representatives of this Agency. We will not permit the deterioration of water quality along the beaches of Long Island or New Jersey, and most certainly will not permit jeopardizing the health, welfare, and well being of individuals using these waters. It must be recognized that moving of the dumping grounds —— sewage sludge and dredge spoil —— at this time, without the availability of environmental background data needed to demonstrate that a new site is suitable or environmentally acceptable, would simply translocate the present problem.

Our Agency with NOAA is moving as quickly as possible towards a better solution of the sludge disposal problem. We are not about to abdicate our responsibilities as individuals, or as an Agency, to such an extent that our position and/or recommendations might cause greater damage to the environment and possibly result in irreparable harm.

APPENDIX A

EPA MONITORING PROGRAM OF SLUDGE/DREDGE SPOIL DUMPING SITES

A comprehensive sampling program was initiated by the EPA, Region II in April 1974 to monitor the quality of water and bottom sediments in the apex of the New York Bight and along the beaches of Long Island and New Jersey. Sites sampled in the program are illustrated in Figure 0 and information as to their location is given in Table 0. This monitoring program is segmented into three phases as shown below:

Type I	Surf Zone	Biweekly
Type II	Near Shore	Monthly
Type III	Transect	Monthly

Type I samples are collected in the surf zone at selected sites along the Long Island and New Jersey coastlines. Type II are collected by boat approximately 100 yards from shore and in water about 15-20 feet deep. Similarly, Type III samples are collected by boat in three transects, each running from the 12-mile sewage sludge dump site, one to the Long Island beaches, a second to the New Jersey beaches, and a third to the New York Harbor entrance. Water-quality data, including sediment characterization, collected at these sampling sites from April to early July 1974 are summarized in Tables 1-12 and selected parameters are illustrated in Figures 1-12.

Long Island Surf Zone and Near Shore

Data from the samples collected in the surf zone and near shore (Figs. 1-2) indicate low total and fecal coliform densities. The levels of fecal coliform at all sampling stations are significantly below the geometric mean density standards for primary and secondary contact recreation waters under New York's Class SB standard of 200 organisms/100 ml. The effect of poorer quality runoff from New York Harbor and through East Rockaway inlet is evident from elevated values observed at LC101 and LC106, respectively. It is important to note that attempts to isolate Salmonella (enteric pathogens) at four sampling stations were unsuccessful.

Data on coliform densities, total organic carbon content, and heavy metals in bottom sediments are illustrated in Figures 3 and 3a for near shore sampling stations. Highest values are again found at stations LIC01 and LIC06, reflecting the influence of New York Harbor and East Rockaway inlet. Comparison of data presented in these two graphs (Figs. 3 and 3a) with similar data collected in the vicinity of the sewage dump site (Figs. 5 and 5a) indicate wide variations in quality. For example, the highest geometric mean fecal coliform recorded at Long Island near shore sampling stations was 400/100g, while that near the dump site was 962/100g. Similarly, along the Long Island near shore the highest mean total organics observed was 2.6 percent and that for metals was 134 mg/kg, while those near the dump site were 4.6 percent and 426 mg/kg, respectively. (Note about 80 mg/kg in the metals value reflect the use of absolute values when summing individual metals reported as less than analytical sensitivity.) Salmonella were not isolated in bottom sediment samples collected at LIC06 even though high fecal coliform densities were observed.

New York Bight Transects

Water and bottom sediment samples were collected along three transects, each originating in the vicinity of the sewage sludge dump site and extending to the Long Island coast, the New Jersey coast, and the New York Harbor entrance.

Geometric mean fecal coliform densities observed in samples collected from the water column in the Long Island transect (Fig. 4) and New York Harbor transect (Fig. 6) indicate that the New York Class SB standard of 200/100 ml is not contravened, except at station NYB30 (436 fecal coliform/100 ml). This station is located closest to New York Harbor and thus, reflects a combination runoff and wastewater discharges from the Hudson River and Raritan Bay. Even at sampling stations within the sewage dump site, observed fecal coliform densities in both surface and bottom water samples were low. A similar review of bacteriological quality in water column samples collected along the New Jersey transect (Fig. 8) indicate that the more stringent New Jersey Class CW-1 standard for primary contact recreation of a geometric mean of 50 fecal coliform/100 ml is not contravened except by the 61/100 ml at the near shore station (NYB20).

Review of the data illustrated for mean coliform densities at bottom sediment sampling stations in the three transects (Figs. 5,7,9) indicate extreme elevated counts of both total and fecal coliform in the vicinity of the sewage sludge dump site. Total coliform densities (geometric means) generally exceed 15,000/100g and fecal coliform, 500/100g in the vicinity of the dump site. A distance of 5-1/2 to 6 miles of low coliform densities separates the high counts in the vicinity of the dump site and the slightly elevated values found at near shore sampling stations. Salmonella were not found at six (6) of the twenty-two (22) transect sampling stations, even though elevated fecal coliform densities were observed at some of the six sites tested.

Results of total organic carbon and heavy metal analyses on bottom sediments at the transect stations (Figs. 5a,7a,9a) also show wide variations in quality. Highest mean values are found in the vicinity of the dump site, with total organic carbon content generally exceeding four percent and metals exceeding 300mg/kg. The data illustrated show a "clean water-sediment" zone of about 5-1/2 to 6 miles separating the dump site from the New Jersey and Long Island bathing beaches. Total organic content are in the main less than one percent and metals under 100/mg/kg in this "clean water-sediment" zone.

New Jersey Surf Zone and Near Shore

Results of sampling in the surf zone and near shore (Figs. 10,12) indicate low total and fecal coliform densities. The level of fecal coliform at all sampling stations generally are far below the geometric mean density standards for primary contact recreation under New Jersey's Class CW-1 standard of 50 organisms/100 ml. Elevated coliform values observed at JC14 (Fig. 10) are related to an ocean outfall from a local municipal treatment plant.

Data on coliform densities, total organic carbon content, and heavy metals in bottom sediments are illustrated in Figures 11 and 11a for near shore sampling stations. Comparison of data presented in these two graphs with similar data collected in the vicinity of the sewage sludge dump site (Figs. 5,5a) indicate wide variations in quality. For example, the highest geometric mean fecal coliform recorded at New Jersey near shore sampling stations was 382/100g, while that near the dump site was 962/100g. Likewise, along the New Jersey near shore the highest mean total organics observed was 1.6 percent and that for metals was 123 mg/kg, while those near the dump site were 4.6 percent and 426 mg/kg, respectively. Note that while tested at two sites, Salmonella were not found in the bottom sediments.

Discussion

Based upon sampling in the surf and near shore waters along the Long Island and New Jersey beaches, it is evident that water quality remains excellent with respect to coliform density and is acceptable for contact recreation. More important, there is no evidence of a trend towards increased coliform density and thus, no indication of degradation. The occasional elevated coliform counts noted in Tables 1,2,10 and 12 appear randomly distributed in time and location, and does not indicate a systematic change or degradation of water quality.

Review of data from sampling in the Bight of the water column and bottom sediments indicate the general locationoof the sludge mass associated with the sewage sludge and dredge spoil dump site. A "clean water-sediment" zone of about 5-1/2 to 6 miles separates athe leading edge of the sludge mass from the New Jersey and Long Island coasts. Slightly elevated organic carbon content and bacterial counts at selected near shore sampling stations can be related to inland occurrences, such as runoff and wastewater discharges.

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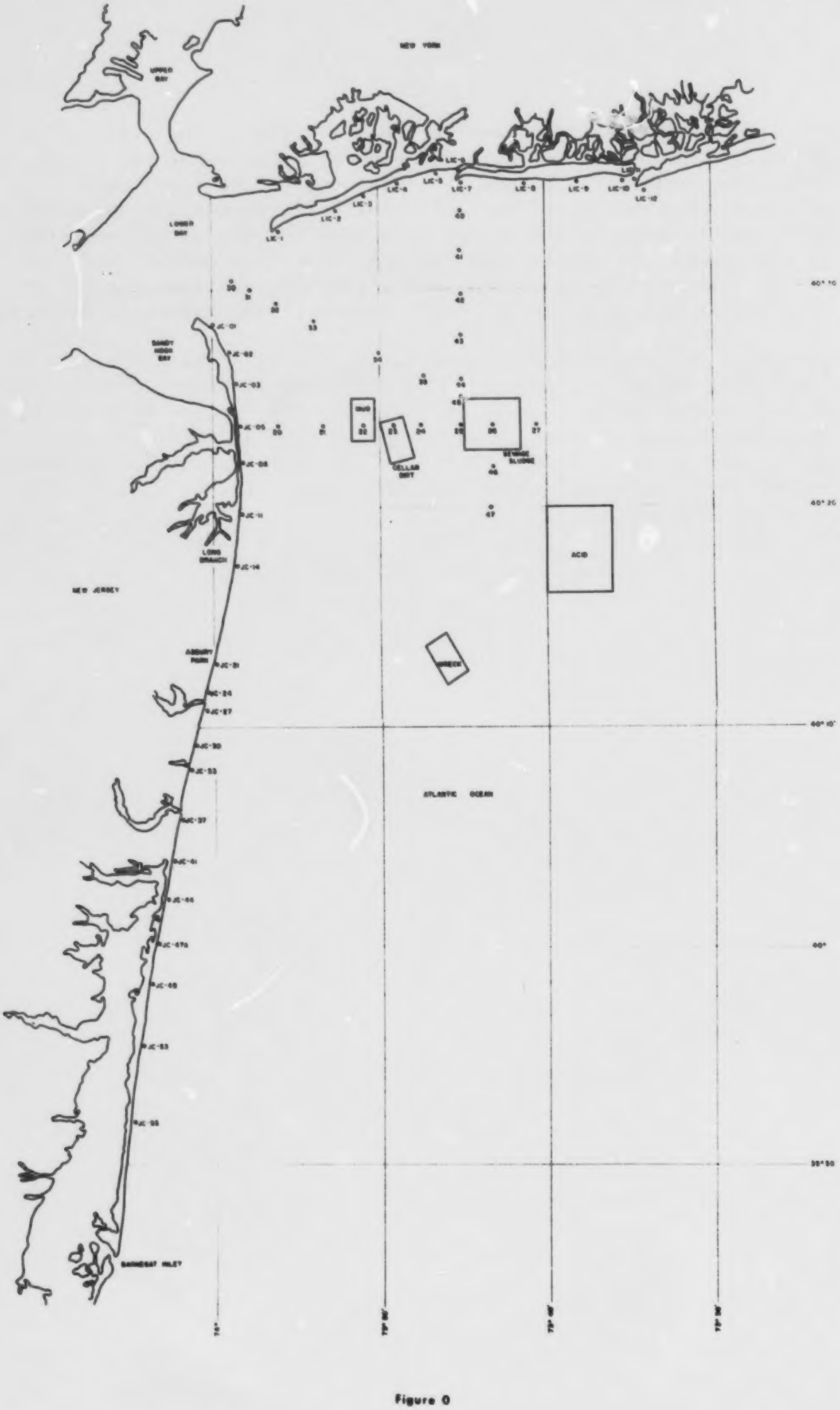
NEW YORK
UPPER BAY
LOWER BAY
SANDY HOOK BAY
LONG BRANCH
NEW JERSEY
ASBURY PARK
BARNEGAT INLET
ATLANTIC OCEAN
DRD
CELLAR DIRT
SEWAGE SLUDGE
ACID
WRECK
LIC-1
LIC-2
LIC-3
LIC-4
LIC-5
LIC-6
LIC-7
LIC-8
LIC-9
LIC-10
LIC-11
LIC-12
NJC-01
NJC-02
NJC-03
NJC-05
NJC-08
NJC-11
NJC-14
NJC-21
NJC-24
NJC-27
NJC-30
NJC-33
NJC-37
NJC-41
NJC-44
NJC-47A
NJC-49
NJC-53
NJC-55
40°30'
40°20'
40°10'
40°
39°50'
74°
73°50'
73°40'
73°30'

Figure 0

TABLE O

STATION LOCATIONS

			Loran		
Station	Latitude	Longitude	3H4 (Brown)	3H5 (Green)	Visuals
NYB20	40°23'54"	73°56'03"	4560	3294	
NYB21	40°23'54"	73°53'30"	4575	3274	
NYB22	40°23'54"	73°51'00"	4590	3253	
NYB23	40°23'54"	73°49'12"	4600	3237	
NYB24	40°23'54"	73°47'30"	4609	3223	
NYB25	40°23'54"	73°45'00"	4624	3200	
NYB26	40°23'54"	73°43'15"	4634	3185	
NYB27	40°23'54"	73°40'32"	4649	3161	
NYB30	40°30'25"	73°58'42"			South of Fl "5"
NYB31	40°30'00"	73°57'36"			South of Fl "3"
NYB32	40°29'25"	73°56'00"			South of Fl "1A" WHIS
NYB33	40°28'36"	73°53'45"			South of BW Mo (A) WHIS
NYB34	40°27'15"	73°50'00"			South of Ambrose Horn
NYB35	40°26'10"	73°47'12"	4629	3220	
NYB40	40°33'36"	73°45'00"	4699	3182	
NYB41	40°31'39"	73°45'00"	4684	3188	
NYB42	40°29'42"	73°45'00"	4669	3193	
NYB43	40°27'45"	73°45'00"	4654	3197	
NYB44	40°25'54"	73°45'00"	4640	3200	
NYB45	40°25'00"	73°45'00"	4633	3200	
NYB46	40°22'00"	73°43'15"	4619	3183	
NYB47	40°20'00"	73°43'15"	4603	3182	

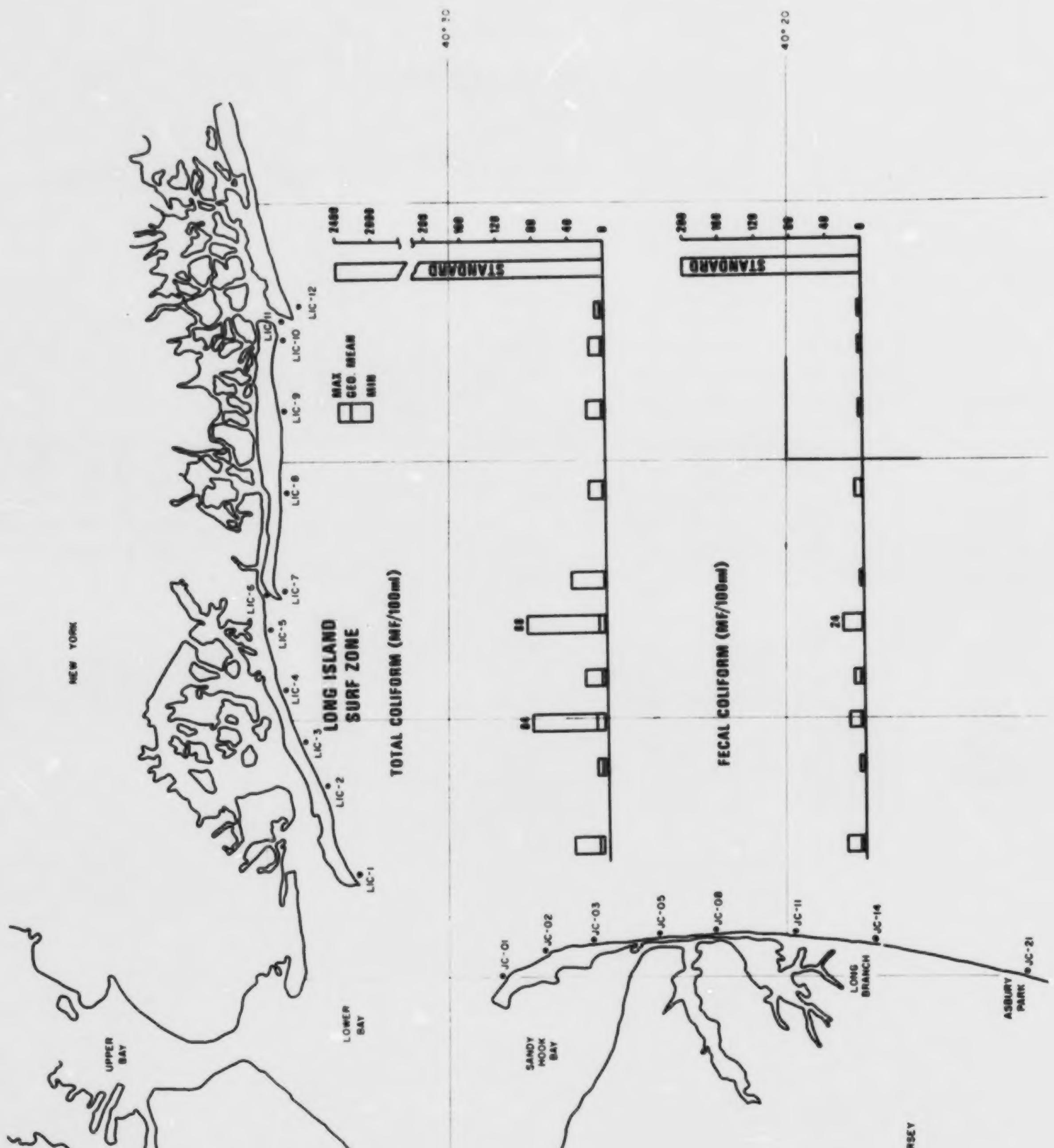
NEW YORK
LONG ISLAND SURF ZONE
TOTAL COLIFORM (MF/100ml)
FECAL COLIFORM (MF/100ml)
MAX
GEO. MEAN
MIN
STANDARD
STANDARD
2400
2000
200
160
120
80
40
0
200
160
120
80
40
0
84
88
24
LIC-1
LIC-2
LIC-3
LIC-4
LIC-5
LIC-6
LIC-7
LIC-8
LIC-9
LIC-10
LIC-11
LIC-12
UPPER BAY
LOWER BAY
SANDY HOOK BAY
JC-01
JC-02
JC-03
JC-05
JC-08
JC-11
JC-14
JC-21
LONG BRANCH
ASBURY PARK
NEW JERSEY
40° 30
40° 20

Figure 1

TABLE 1

SUMMARY OF BACTERIOLOGICAL DATA (WATER COLUMN) COLLECTED ALONG LONG ISLAND SHORELINE

(Samples Collected in Surf Zone)

Date	Station Identification Number									
	LIC01		LIC02		LIC03		LIC04		LIC05	
	TC	FC	TC	FC	TC	FC	TC	FC	TC	FC
4/11/74	38	20	11	6	7	7	1	1	<1	<1
4/23/74	10	7	11	5	84	17	19	10	88	24
5/7/74	5	1	6	3	2	<1	4	1	2	1
5/22/74	12	5	3	2	3	2	9	6	48	3
6/5/74	8	3	4	1	32	11	25	3	5	2
6/19/74	--	--	2	<1	44	5	3	<1	4	<1
7/2/74	8	6	12	<1	5	<1	4	<1	8	<1
Samples to date	6	6	7	7	7	7	7	7	7	7
Median	9		6		7		4		5	
Geometric Mean	11	4	6	2	12	4	5	2	7	2
Max	38	20	12	6	84	17	25	10	88	24
Min	5	1	2	<1	2	<1	1	<1	<1	<1

TC = Total Coliform (MF/100 ml)
FC = Fecal Coliform (MF/100 ml)
< = Less than

TABLE 1 - Cont.

SUMMARY OF BACTERIOLOGICAL DATA (WATER COLUMN) COLLECTED ALONG LONG ISLAND SHORELINE

(Samples Collected in Surf Zone)

Date	Station Identification Number									
	LIC07		LIC08		LIC09		LIC10		LIC12	
	TC	FC	TC	FC	TC	FC	TC	FC	TC	FC
4/11/74	<1	<1	4	1	1	<1	7	3	<1	<1
4/23/74	8	3	11	8	10	4	3	1	7	3
5/7/74	<1	<1	1	<1	2	<1	1	<1	1	<1
5/22/74	15	<1	2	1	5	1	16	3	8	1
6/5/74	<1	<1	1	1	2	<1	2	2	--	--
6/19/74	39	4	18	1	8	4	9	3	7	3
7/2/74	4	<1	2	<1	2	<1	2	<1	4	<1
Samples to date	7	7	7	7	7	7	7	7	6	6
Median	<1		2		2		3		7	
Geometric Mean	2	1	3	1	3	1	3	2	3	2
Max	39	4	18	8	10	<1	16	3	8	3
Min	<1	<1	1	<1	1	4	1	<1	<1	<1

TC = Total Coliform (MF/100 ml)
FC = Fecal Coliform (MF/100 ml)
< = Less than

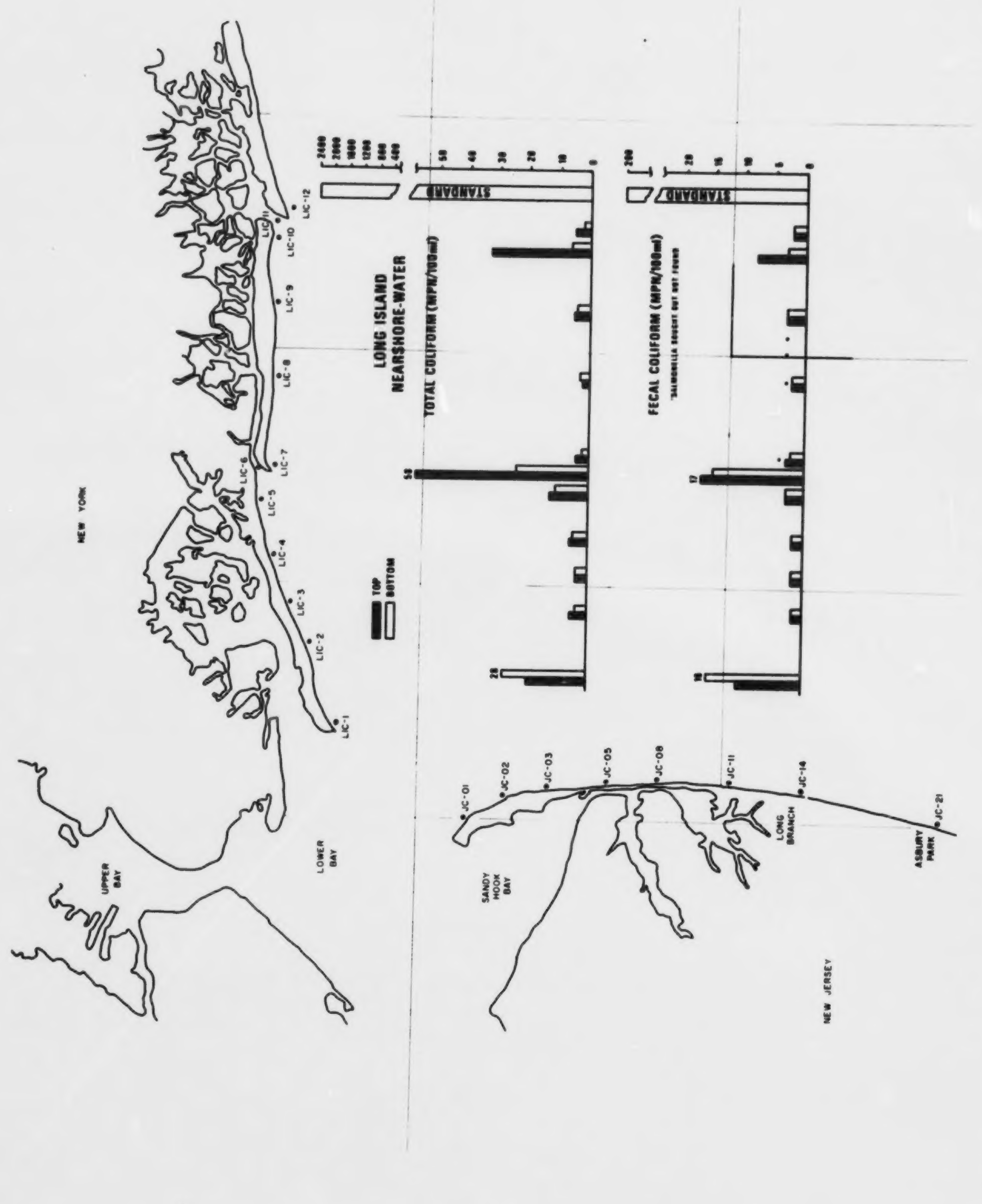
LONG ISLAND
NEARSHORE-WATER
TOTAL COLIFORM (MPN/100ml)
FECAL COLIFORM (MPN/100ml)
*SALMONELLA SOUGHT BUT NOT FOUND
TOP
BOTTOM
STANDARD
2400
2000
1600
1200
800
400
50
40
30
20
10
0
200
20
15
10
5
0
28
58
16
17
NEW YORK
UPPER BAY
LOWER BAY
SANDY HOOK BAY
NEW JERSEY
LONG BRANCH
ASBURY PARK
LIC-1
LIC-2
LIC-3
LIC-4
LIC-5
LIC-6
LIC-7
LIC-8
LIC-9
LIC-10
LIC-11
LIC-12
JC-01
JC-02
JC-03
JC-05
JC-08
JC-11
JC-14
JC-21
40°30
40°20

Figure 2

TABLE 2

SUMMARY OF WATER-QUALITY DATA (WATER COLUMN) COLLECTED ALONG THE LONG ISLAND COAST
(by boat approximately 100 yards from beach)

			Station Identification Number										
Parameter	Date	Top/ Bottom	LIC01	LIC02	LIC03	LIC04	LIC05	LIC06	LIC07	LIC08	LIC09	LIC11	LIC12
Total Coliform (MPN/100 ml)	05-01-74	T	790	23	2	7	8	3,480	2	< 2	< 2	1,300	31
	05-01-74	B	2,210	< 2	< 2	11	17	3,480	< 2	< 2	2	8	< 2
	06-06-74	T	5	4	7	7	17	8	17	2	5	4	< 2
	06-06-74	B	2	2	5	2	17	< 2	< 2	5	2	11	< 2
	07-11-74	T	2	< 2	5	5	17	7	2	< 2	13	7	< 2
	07-11-74	B	5	11	7	5	5	2	2	2	17	< 2	2
Fecal Coliform (MPN/100 ml)	05-01-74	T	330	2	< 2	< 2	< 2	1,300	2	< 2	< 2	109	< 2
	05-01-74	B	1,090	< 2	< 2	< 2	< 2	790	< 2	< 2	< 2	5	< 2
	06-06-74	T	< 2	< 2	2	< 2	2	< 2	4	< 2	2	2	< 2
	06-06-74	B	< 2	< 2	< 2	< 2	5	< 2	< 2	2	2	4	< 2
	07-11-74	T	< 2	< 2	2	2	5	< 2	< 2	< 2	5	2	< 2
	07-11-74	B	< 2	< 2	4	< 2	< 2	< 2	< 2	2	4	< 2	2
Water Temperature (°C)	05-01-74	T	9.4	9.0	9.8	9.6	10.7	11.5	9.3	10.1	9.6	8.9	
	05-01-74	B	8.1	8.5	8.7	8.2	9.5	11.5	8.8	9.9	8.4	7.6	
	06-06-74	T	16.4	16.8	16.6	16.9	15.8	15.8	16.3	16.8	17.9	15.7	16.2
	06-06-74	B	16.2	15.2	16.6	16.4	15.2	15.6	15.6	16.7	17.4	15.7	15.4
	07-11-74	T	16.1	15.8	15.4	15.2	16.1	15.2	14.7	15.2	18.3	16.3	16.0
	07-11-74	B	16.1	15.7	15.2	15.1	14.8	15.3	14.3	14.7	16.0	15.5	15.5
Dissolved Oxygen (mg/l)	05-01-74	T	9.4	9.4	10.7	9.6	9.6	7.8	10.3	9.6	9.6	10.0	8.5
	05-01-74	B	8.9	9.3	10.5	8.9	9.2	7.8	10.2	9.7	9.0	9.2	9.5
	06-06-74	T	8.2	8.4	8.3	8.9	8.3	8.4	8.4	9.2	9.6	8.6	9.5
	06-06-74	B	8.4	8.0	7.8	7.2	7.4	8.0	7.2	9.3	9.1	8.6	8.6
	07-11-74	T	5.7	6.0	4.8	4.5	5.2	4.7	4.7	5.0	6.9	6.0	6.0
	07-11-74	B	5.7	5.9	4.8	4.3	4.3	4.6	4.3	4.4	6.0	6.0	5.8
Conductivity (micromhos)	05-01-74	T	33,000	33,100	33,400	34,000	34,200	34,900	33,700	34,500	34,200	33,500	
	05-01-74	B	32,600	33,000	33,200	33,100	33,600	34,800	33,500	34,400	33,400	32,600	
	06-06-74	T	35,600	35,900	35,900	36,300	36,600	36,000	36,400	36,900	37,500	35,800	36,300
	06-06-74	B	35,600	34,900	35,700	36,200	35,000	35,600	36,100	36,700	37,400	35,800	35,400
	07-11-74	T	37,000	36,300	36,400	36,300	36,900	36,500	36,100	36,300	38,600	37,200	37,300
	07-11-74	B	37,000	36,800	36,400	36,200	36,200	36,500	35,900	35,900	37,100	36,900	36,900
Salinity (g/l)	05-01-74	T	29.6	29.9	29.7	30.5	29.9	30.0	30.6	30.6	30.8	30.6	
	05-01-74	B	30.4	30.4	30.4	31.0	30.2	29.9	30.5	30.8	31.1	30.9	
	06-06-74	T	27.2	27.2	27.3	27.4	27.3	28.0	27.9	28.0	27.8	27.7	27.9
	06-06-74	B	27.2	27.4	27.1	27.5	27.5	27.6	28.1	28.0	28.0	27.8	27.4
	07-11-74	T	28.7	28.2	28.5	28.6	28.6	28.7	28.7	28.7	28.5	28.8	28.9
	07-11-74	B	28.9	28.8	28.8	28.6	29.0	28.7	28.9	28.7	28.8	29.0	29.9

T Top - Approximately 2 ft. from surface
B Bottom - Approximately 2 ft. from bottom (Total depth at sampling sites ranged from 12-22 ft.)

LIC06 E. Rockaway Inlet
LIC11 Jones Inlet

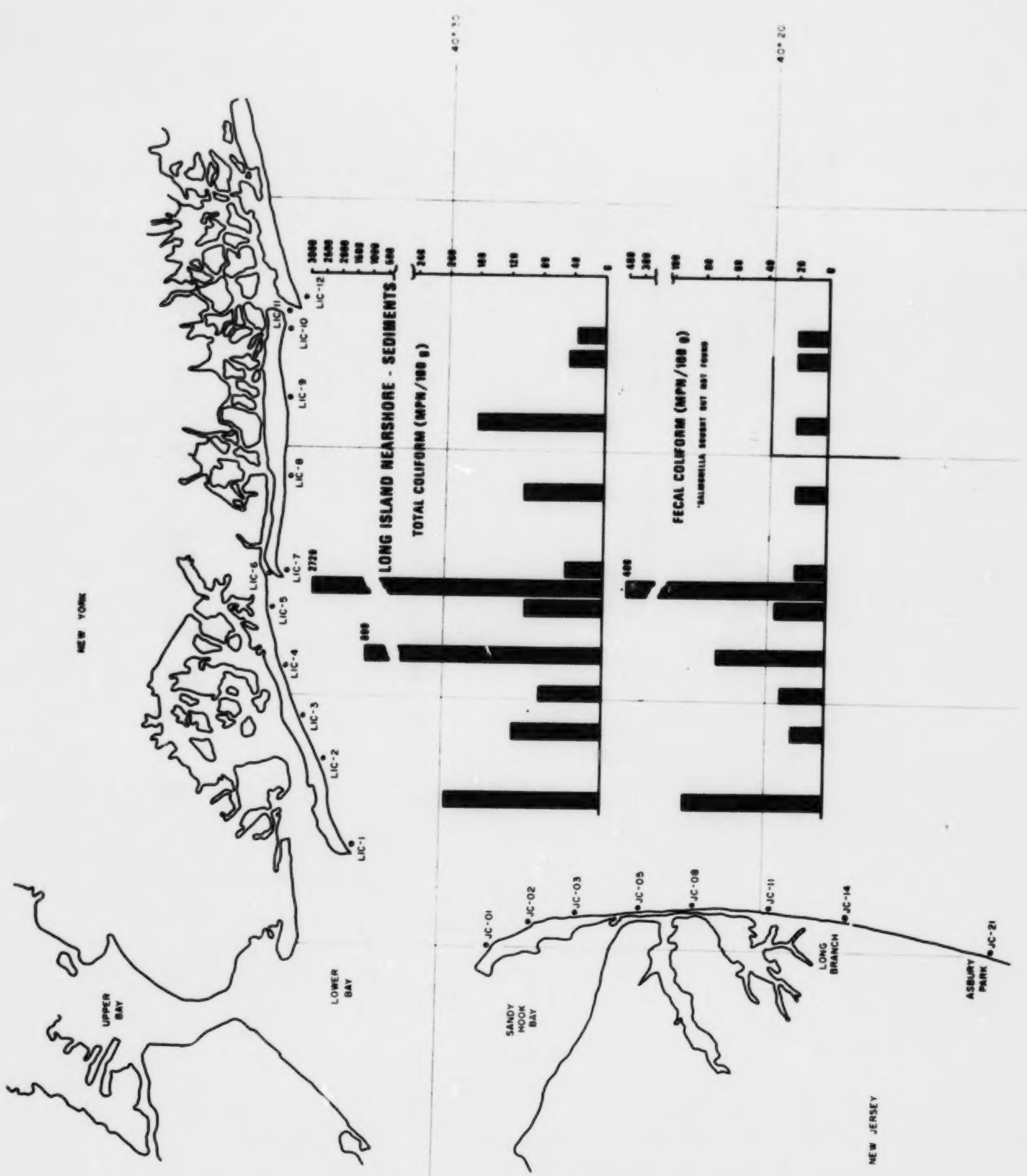
LONG ISLAND NEARSHORE - SEDIMENTS
TOTAL COLIFORM (MPN/100 g)
FECAL COLIFORM (MPN/100 g)
*SALMONELLA SOUGHT BUT NOT FOUND
0
40
80
120
160
200
240
500
1000
1500
2000
2500
3000
2720
800
400
200
400
0
20
40
60
80
100
NEW YORK
UPPER BAY
LOWER BAY
SANDY HOOK BAY
NEW JERSEY
LONG BRANCH
ASBURY PARK
LIC-1
LIC-2
LIC-3
LIC-4
LIC-5
LIC-6
LIC-7
LIC-8
LIC-9
LIC-10
LIC-11
LIC-12
JC-01
JC-02
JC-03
JC-05
JC-08
JC-11
JC-14
JC-21
40° 30
40° 20

Figure 3

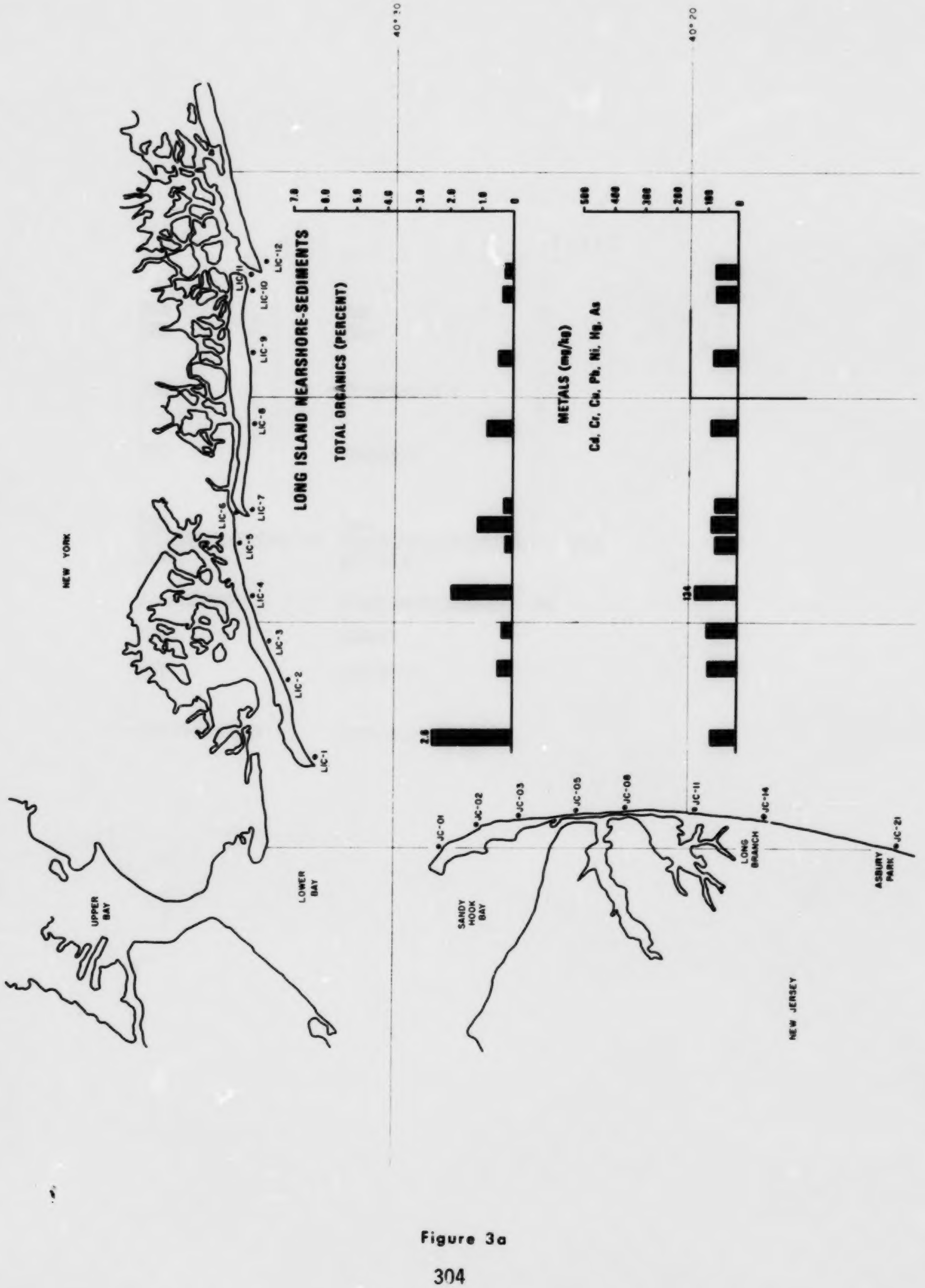
LONG ISLAND NEARSHORE-SEDIMENTS
TOTAL ORGANICS (PERCENT)
METALS (mg/kg)
Cd, Cr, Cu, Pb, Ni, Hg, As
7.0
6.0
5.0
4.0
3.0
2.0
1.0
0
500
400
300
200
100
0
2.8
134
40° 30
40° 20
NEW YORK
UPPER BAY
LOWER BAY
SANDY HOOK BAY
LONG BRANCH
ASBURY PARK
NEW JERSEY
LIC-1
LIC-2
LIC-3
LIC-4
LIC-5
LIC-6
LIC-7
LIC-8
LIC-9
LIC-10
LIC-11
LIC-12
JC-01
JC-02
JC-03
JC-05
JC-08
JC-11
JC-14
JC-21

Figure 3a

TABLE 3

SUMMARY OF WATER-QUALITY DATA (BOTTOM SEDIMENTS) COLLECTED ALONG THE LONG ISLAND COAST
(Samples collected by boat approximately 100 yds. from beach)

Parameter	Date	Station Identification Number											
		LIC01	LIC02	LIC03	LIC04	LIC05	LIC06	LIC07	LIC08	LIC09	LIC10	LIC11	LIC12
Total Coliform (MPN/100 g)	05-01-74	7,900	130	220	490	460	2,210	70	230	230	--	230	110
	06-06-74	< 20	230	50	13,000	110	7,000	80	20	80	--	< 20	20
	07-11-74	50	50	50	80	20	1,300	20	230	230	--	< 20	20
Fecal Coliform (MPN/100 g)	05-01-74	1,720	< 20	50	50	80	630	< 20	< 20	< 20	--	< 20	20
	06-06-74	< 20	20	< 20	330	< 20	790	< 20	< 20	< 20	--	< 20	< 20
	07-11-74	< 20	20	20	< 20	< 20	130	< 20	< 20	< 20	--	< 20	< 20
Salmonella (qualitative)	05-01-74	--	--	--	--	--	Neg.	--	--	--	--	--	--
Total Organics (mg/kg)	05-01-74	--	5,450	2,510	3,850	1,770	2,570	3,300	10,600	5,620	--	1,240	3,000
	06-06-74	25,600	3,400	3,860	34,100	2,430	18,300	1,810	4,900	3,090	--	1,280	1,840
	07-11-74	2,550	12,770	2,490	5,260	1,450	2,890	1,980	3,340	3,340	--	1,450	2,990
Cadmium (mg/kg)	05-01-74	--	--	--	--	< 5	< 5	< 5	< 5	--	--	< 5	< 5
	06-06-74	< 3	< 3	< 3	< 3	--	--	--	--	< 3	--	--	--
	07-11-74	< 5	< 5	< 5	< 5	< 5	< 5	< 5	< 5	< 5	--	< 5	< 5
Chromium (mg/kg)	05-01-74	--	--	--	--	4	< 3	4	12	--	--	< 3	5
	06-06-74	5	20	20	40	--	--	--	--	< 4	--	--	--
	07-11-74	< 10	< 10	--	< 10	< 10	< 10	< 10	10	< 10	--	< 10	< 10
Copper (mg/kg)	05-01-74	--	--	--	--	< 3	3	3	15	--	--	< 3	< 3
	06-06-74	< 6	< 6	< 6	24	--	--	--	--	< 6	--	--	--
	07-11-74	5	4	4	< 3	< 3	4	< 3	4	3	--	3	3
Lead (mg/kg)	05-01-74	--	--	--	--	< 40	< 40	< 40	< 40	--	--	< 40	< 40
	06-06-74	< 50	< 50	< 50	80	--	--	--	--	< 50	--	--	--
	07-11-74	< 50	< 50	< 50	< 50	< 50	< 50	< 50	< 50	< 50	--	< 50	< 50
Nickel (mg/kg)	05-01-74	--	--	--	--	< 10	< 10	< 10	< 10	--	--	< 10	< 10
	06-06-74	< 10	< 10	< 10	< 10	--	--	--	--	< 10	--	--	--
	07-11-74	< 10	< 10	< 10	< 10	< 10	< 10	< 10	15	< 10	--	< 10	< 10
Mercury (mg/kg)	05-01-74	--	--	--	--	0.1	0.5	< 0.1	< 0.1	--	--	< 0.1	< 0.1
	06-06-74	< .5	< .5	< .5	1.02	--	--	--	--	< .5	--	--	--
	07-11-74	.16	.12	.08	.14	.12	.07	.08	.21	.52	--	.01	.21
Arsenic (mg/kg)	05-01-74	--	--	--	--	.8	.9	1.8	2.4	--	--	.4	1.2
	06-06-74	< .5	4.3	1.3	16	--	--	--	--	0.7	--	--	--
	07-11-74	.6	.9	.7	.6	.5	< .5	.6	< .5	.6	--	< .5	.6

Neg. - Sought but not detected

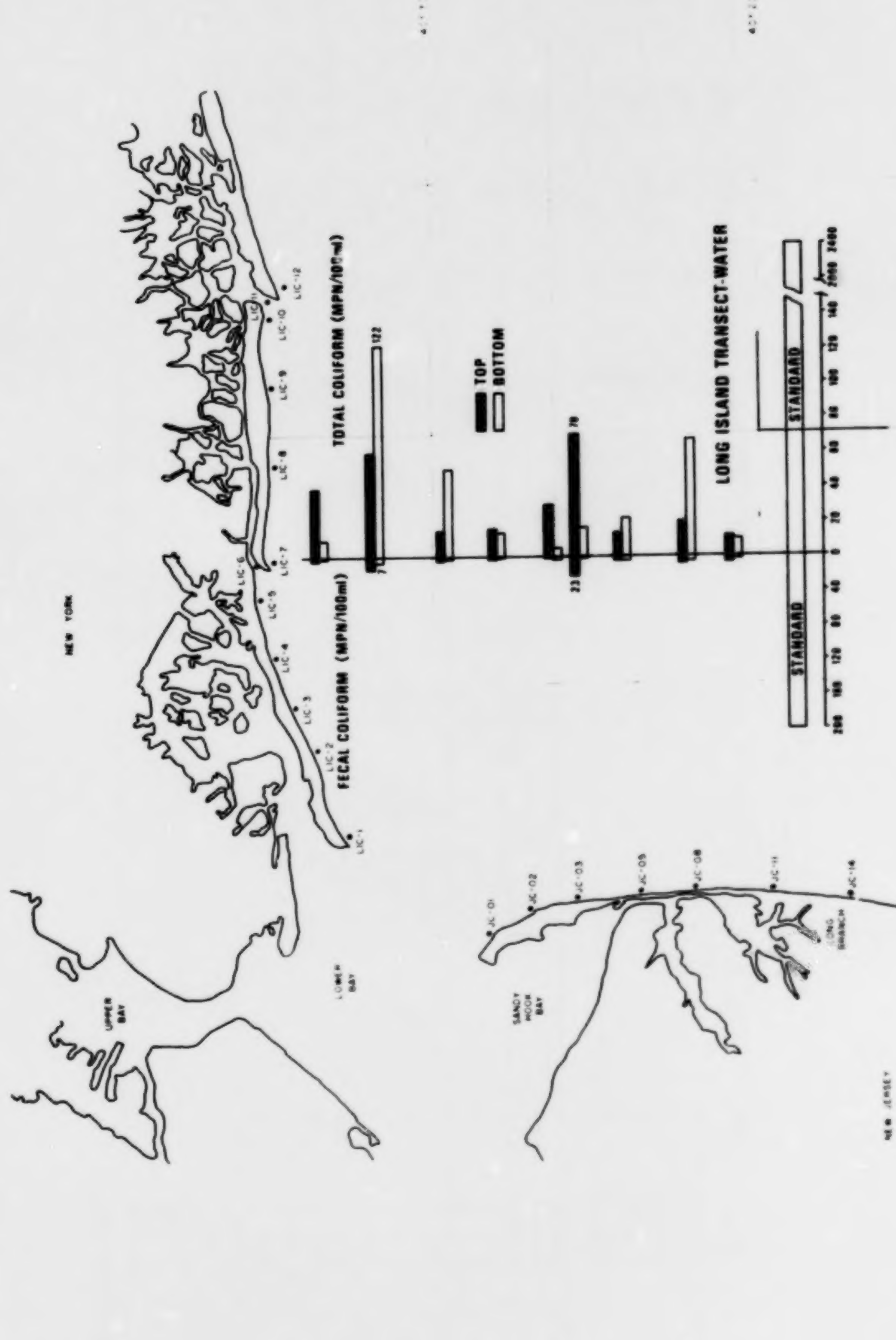
TOTAL COLIFORM (MPN/100ml)
FECAL COLIFORM (MPN/100ml)
TOP
BOTTOM
LONG ISLAND TRANSECT-WATER
STANDARD
STANDARD
122
71
78
23
200
160
120
80
40
0
20
40
60
80
100
120
140
2000
2400
NEW YORK
NEW JERSEY
UPPER BAY
LOWER BAY
SANDY HOOK BAY
LONG BRANCH
ASBURY PARK
LIC-1
LIC-2
LIC-3
LIC-4
LIC-5
LIC-6
LIC-7
LIC-8
LIC-9
LIC-10
LIC-11
LIC-12
JC-01
JC-02
JC-03
JC-05
JC-08
JC-11
JC-14
JC-21

Figure 4

TABLE 4

SUMMARY OF WATER-QUALITY DATA (WATER COLUMN) COLLECTED ALONG LONG ISLAND TRANSECT

Parameter	Date	Top/ Bottom	Station Identification Number							
			NYB40	NYB41	NYB42	NYB43	NYB44	NYB45	NYB46	NYB47
Total Coliform (MPN/100 ml)	04-17-74	T	13+	2+	< 2+	--	--	5,420*	2,210	420
	04-17-74	B	109+	700+	230+	--	--	49*	8	7
	05-14-74	T	330	3,480	79	5	11	8	< 2$	< 2$
	05-14-74	B	2	1,300	278	79	14	49	79$	17$
	07-09-74	T	13	8	23	49	79	8	< 2#	< 2#
	07-09-74	B	< 2	< 2	< 2	< 2	< 2	< 2	490#	8#
Fecal Coliform (MPN/100 ml)	04-17-74	T	< 2+	< 2+	< 2+	--	--	1,300*	49	49
	04-17-74	B	5+	13+	8+	--	--	< 2*	2	< 2
	05-14-74	T	27	700	11	< 2	< 2	5	< 2$	< 2$
	05-14-74	B	< 2	13	2	5	5	< 2	5$	< 2$
	07-09-74	T	< 2	< 2	5	8	2	< 2	< 2#	< 2#
	07-09-74	B	< 2	< 2	< 2	< 2	< 2	< 2	11#	< 2#
Water Temperature (°C)	04-17-74	T	--	--	--	--	--	7.7*	6.9	6.8
	04-17-74	B	--	--	--	--	--	6.7*	7.0	6.5
	05-14-74	T	12.5	12.1	12.1	12.2	11.7	11.7	14.1$	13.9$
	05-14-74	B	10.0	11.0	11.7	11.8	11.7	11.6	--$	--$
	07-09-74	T	19.4	19.5	20.1	19.9	20.7	21.0	17.5#	17.3#
	07-09-74	B	14.2	13.2	13.9	13.5	14.8	15.1	10.9#	12.1#
Dissolved Oxygen (mg/1)	04-17-74	T	--	--	--	--	--	11.0*	7.3	10.9
	04-17-74	B	--	--	--	--	--	9.2*	8.6	10.0
	05-14-74	T	7.2	7.4	5.6	5.9	3.7	2.7	9.7$	10.3$
	05-14-74	B	5.0	3.5	5.4	4.0	3.4	2.3	7.6$	7.6$
	07-09-74	T	--	--	--	--	--	--	9.0#	8.8#
	07-09-74	B	--	--	--	--	--	--	6.2#	4.7#
Conductivity (micromhos)	04-17-74	T	--	--	--	--	--	30,700*	29,300	28,600
	04-17-74	B	--	--	--	--	--	31,400*	31,000	30,600
	05-14-74	T	32,400	32,300	33,400	33,200	34,100	34,200	35,200$	34,700$
	05-14-74	B	34,300	34,400	34,300	34,400	34,700	34,800	--$	--$
	07-09-74	T	38,500	37,100	37,500	36,100	36,500	37,200	37,500#	37,800#
	07-09-74	B	35,800	34,600	34,800	34,500	35,700	36,000	34,700#	33,000#
Salinity (g/l)	04-17-74	T	--	--	--	--	--	28.5*	27.8	27.0
	04-17-74	B	--	--	--	--	--	30.3*	29.7	29.0
	05-14-74	T	--	27.8	27.9	27.7	28.8	29.1	28.2$	27.7$
	05-14-74	B	--	29.6	28.8	29.1	29.5	29.6	--$	--$
	07-09-74	T	28.2	26.6	26.5	25.7	25.5	26.2	28.3#	28.4#
	07-09-74	B	28.8	28.6	28.7	28.3	29.0	28.8	28.5#	28.1#

T Top - Approximately 2 ft. below surface
B Bottom - Approximately 2 ft. above bottom

* Samples Collected 04-18-74
+ Samples Collected 04-24-74
$ Samples Collected 05-21-74
# Samples Collected 06-14-74

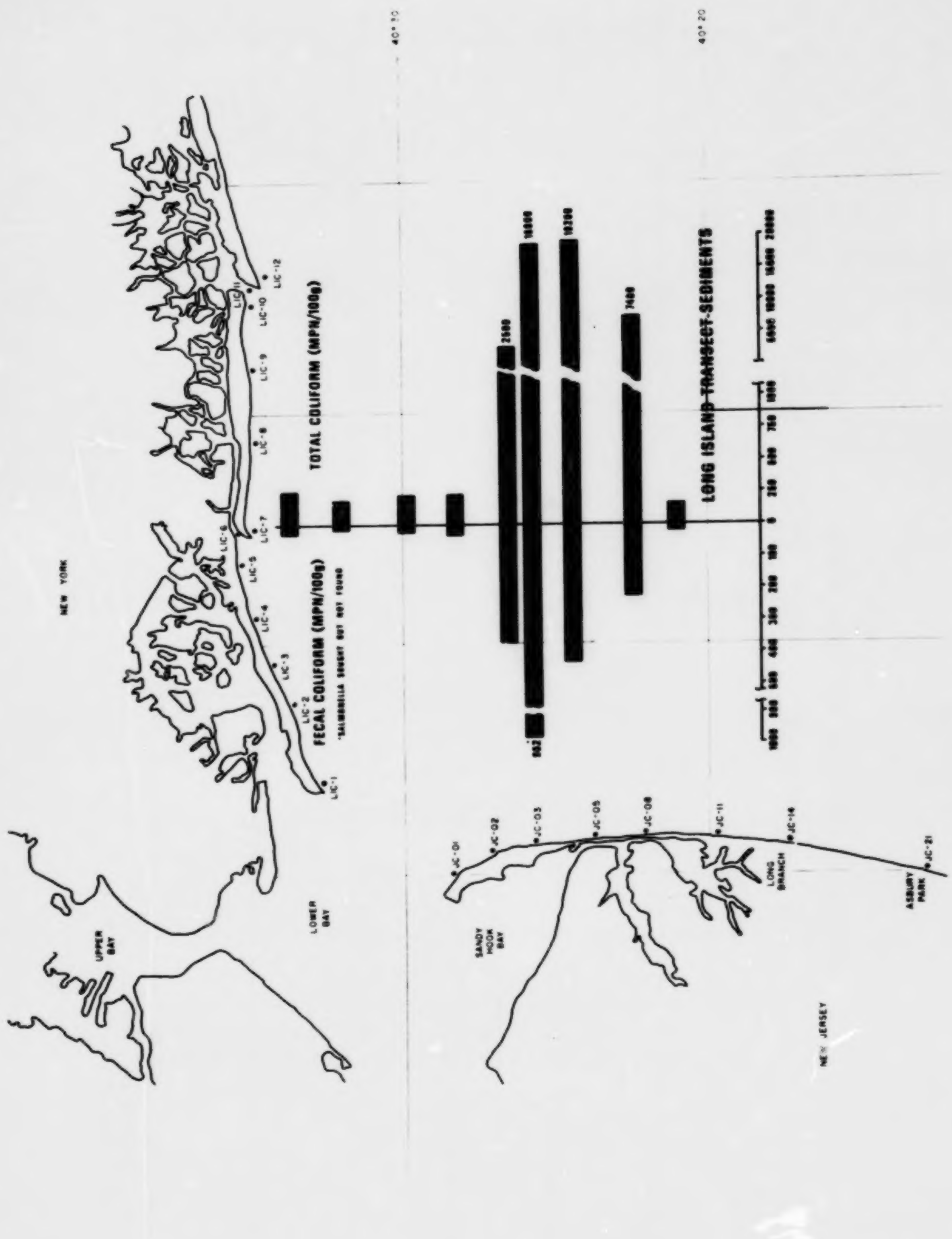
NEW YORK
UPPER BAY
LOWER BAY
LIC-1
LIC-2
LIC-3
LIC-4
LIC-5
LIC-6
LIC-7
LIC-8
LIC-9
LIC-10
LIC-11
LIC-12
FECAL COLIFORM (MPN/100g)
*SALMONELLA SOUGHT BUT NOT FOUND
TOTAL COLIFORM (MPN/100g)
40° 30
40° 20
SANDY HOOK BAY
JC-01
JC-02
JC-03
JC-05
JC-08
JC-11
JC-14
JC-21
LONG BRANCH
ASBURY PARK
NEW JERSEY
2500
902*
18000
19200
7400
LONG ISLAND TRANSECT-SEDIMENTS
1000
900
500
400
300
200
100
0
250
500
750
1000
5000
10000
15000
20000

Figure 5

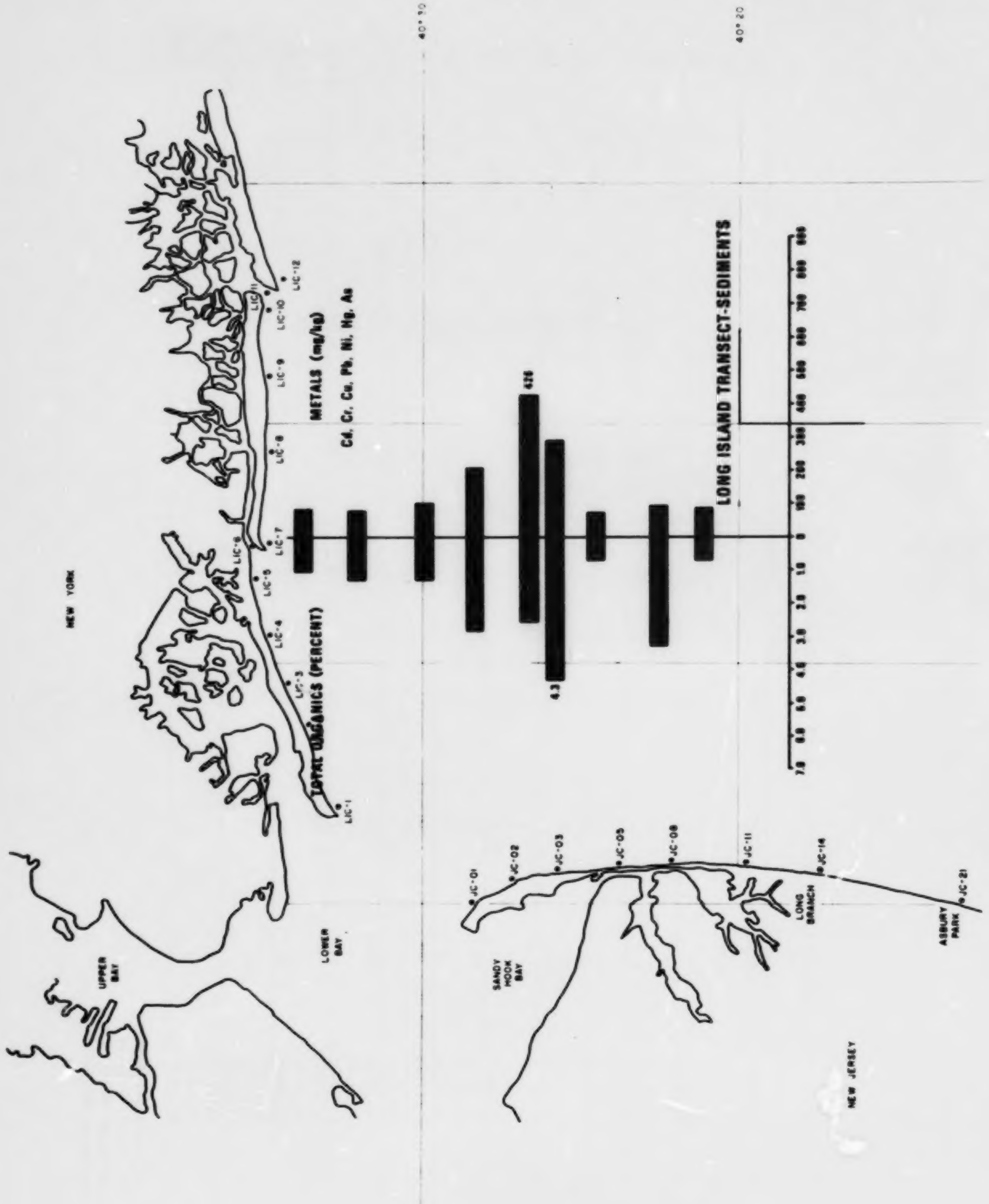
NEW YORK
UPPER BAY
LOWER BAY
SANDY HOOK BAY
LONG BRANCH
ASBURY PARK
NEW JERSEY
LIC-1
LIC-3
LIC-4
LIC-5
LIC-6
LIC-7
LIC-8
LIC-9
LIC-10
LIC-11
LIC-12
JC-01
JC-02
JC-03
JC-05
JC-08
JC-11
JC-14
JC-21
40° 30
40° 20
TOTAL ORGANICS (PERCENT)
METALS (mg/kg)
Cd, Cr, Cu, Pb, Ni, Hg, As
426
4.3
LONG ISLAND TRANSECT-SEDIMENTS
7.0
6.0
5.0
4.0
3.0
2.0
1.0
0
100
200
300
400
500
600
700
800
900

Figure 5a

TABLE 5

SUMMARY OF WATER-QUALITY DATA (BOTTOM SEDIMENTS) COLLECTED ALONG LONG ISLAND TRANSECT

Parameter	Date	Station Identification Number							
		NYB40	NYB41	NYB42	NYB43	NYB44	NYB45	NYB46	NYB47
Total Coliform (MPN/100 g)	04-17-74	330+	270+	--	--	--	34,800*	2,300	490
	05-14-74	490	230	2,780	2,300	1,300	17,500	7,900$	130$
	07-09-74	110	130	< 20	20	4,900	10,900	22,100#	70#
Fecal Coliform (MPN/100 g)	04-17-74	< 20+	40+	--	--	--	790*	50	20
	05-14-74	50	< 20	20	50	170	230	230$	< 20$
	07-09-74	< 20	< 20	< 20	20	790	4,900	940#	20#
Salmonella (qualitative)	05-14-74	--	--	--	--	--	Neg.	--	--
	07-09-74	--	--	--	--	Neg.	--	--	--
Total Organics (mg/kg)	04-17-74	17,500+	7,320+	--	--	--	48,700*	7,560	7,630
	05-14-74	5,890	16,100	15,000	28,800	21,400	75,500	76,600$	6,840$
	07-09-74	8,370	14,700	10,700	27,600	29,600	4,660	13,700#	6,770#
Cadmium (mg/kg)	04-17-74	< 5+	< 5+	--	--	--	9*	< 3	< 3
	05-14-74	--	--	< 2	2	< 2	2	--	--
	07-09-74	< 5	< 5	< 5	< 5	9	< 5	< 3#	< 3#
Chromium (mg/kg)	04-17-74	13+	9+	--	--	--	115*	9	8
	05-14-74	--	--	12	54	30	48	--	--
	07-09-74	12	< 10	19	71	146	12	10#	8#
Copper (mg/kg)	04-17-74	4+	4+	--	--	--	133*	5	4
	05-14-74	--	--	9	42	38	97	--	--
	07-09-74	5	< 3	8	51	156	8	10#	19#
Lead (mg/kg)	04-17-74	< 40+	< 40+	--	--	--	208*	< 40	< 40
	05-14-74	--	--	< 50	82	50	130	--	--
	07-09-74	< 50	< 50	< 50	60	370	< 50	< 50#	< 50#
Nickel (mg/kg)	04-17-74	< 10+	< 10+	--	--	--	17*	6	5
	05-14-74	--	--	7	11	9	12	--	--
	07-09-74	< 10	< 10	< 10	15	27	< 10	< 10#	< 10#
Mercury (mg/kg)	04-17-74	.2+	< .2+	--	--	--	1.8*	5.0	1.4
	05-14-74	--	--	.2	.8	.7	1.1	--	1.0$
	07-09-74	.42	.06	.09	.65	2.4	1.35	.16#	< .1#
Arsenic (mg/kg)	04-17-74	6.3+	5.2+	--	--	--	3.7*	5.6	3.0
	05-14-74	--	--	17.8	7.8	6.3	4.2	--	--
	07-09-74	2.4	5.2	26.0	4.8	3.6	.8	21#	2.2#

* Samples Collected 04-18-74
+ Samples Collected 04-24-74
$ Samples Collected 05-21-74
# Samples Collected 06-14-74

Neg. - Sought but not detected

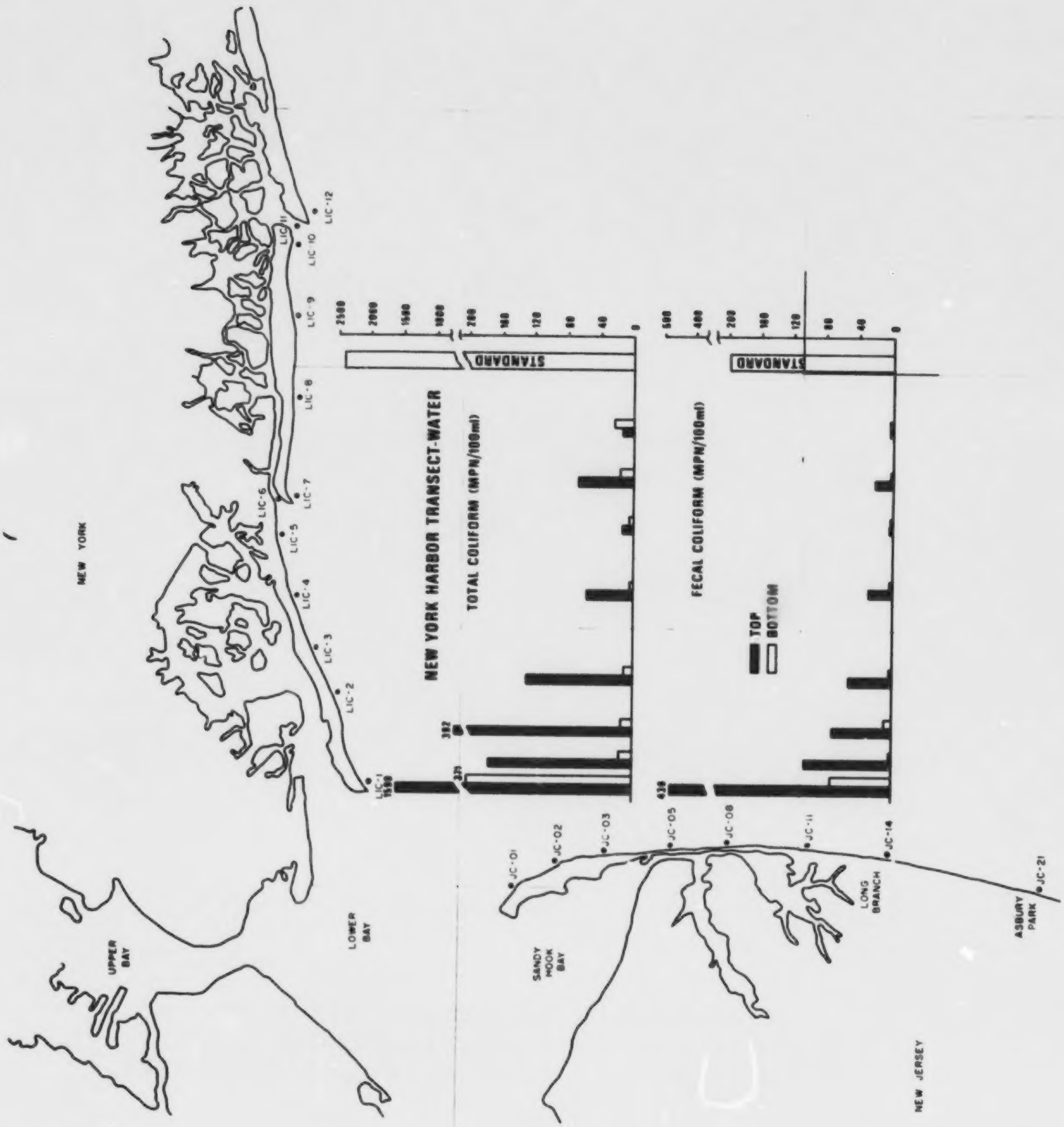
NEW YORK
UPPER BAY
LOWER BAY
SANDY HOOK BAY
NEW JERSEY
LONG BRANCH
ASBURY PARK
LIC-1
LIC-2
LIC-3
LIC-4
LIC-5
LIC-6
LIC-7
LIC-8
LIC-9
LIC-10
LIC-11
LIC-12
JC-01
JC-02
JC-03
JC-05
JC-08
JC-11
JC-14
JC-21
NEW YORK HARBOR TRANSECT-WATER
TOTAL COLIFORM (MPN/100ml)
FECAL COLIFORM (MPN/100ml)
TOP
BOTTOM
STANDARD
STANDARD
2500
2000
1500
1000
200
160
120
80
40
0
500
400
200
160
120
80
40
0
1500
225
392
420

Figure 6

TABLE 6

SUMMARY OF WATER-QUALITY DATA (WATER COLUMN) COLLECTED ALONG N.Y. HARBOR TRANSECT

Parameter	Date	Top/ Bottom	Station Identification Number					
			NYB30	NYB31	NYB32	NYB33	NYB34	NYB35
Total Coliform (MPN/100 ml)	04-17-74	T	3,480	--	2,400	3,480	1,300	130
	04-17-74	B	330	--	23	22	< 2	2
	05-21-74	T	330	172	109	< 2	< 2	< 2
	05-21-74	B	70	17	33	13	2	17
	07-09-74	T	3,480*	--	230	330	79	11+
	07-09-74	B	2,210*	--	5	8	8	2+
Fecal Coliform (MPN/100 ml)	04-17-74	T	490	--	790	1,090	230	9
	04-17-74	B	109	--	23	< 2	< 2	< 2
	05-21-74	T	130	109	23	< 2	< 2	< 2
	05-21-74	B	5	< 2	17	< 2	2	2
	07-09-74	T	1,300*	--	23	79	8	2+
	07-09-74	B	790*	--	2	5	2	< 2+
Water Temperature (^oC)	04-17-74	T	7.5	--	7.1	7.2	7.9	8.0
	04-17-74	B	6.4	--	5.8	5.9	6.5	7.4
	05-21-74	T	11.9	12.3	12.7	12.8	13.0	14.2
	05-21-74	B	10.2	9.6	10.8	9.5	--	--
	07-09-74	T	18.6*	--	21.3	19.4	19.6	17.6+
	07-09-74	B	16.7*	--	14.0	13.0	10.9	18.2+
Dissolved Oxygen (mg/1)	04-17-74	T	9.8	--	10.9	8.5	10.0	11.9
	04-17-74	B	7.2	--	9.3	9.3	9.1	9.5
	05-21-74	T	9.0	9.4	9.6	8.7	9.9	9.5
	05-21-74	B	7.3	9.4	8.2	6.3	6.7	6.5
	07-09-74	T	7.2*	--	--	--	--	9.5+
	07-09-74	B	7.0*	--	--	--	--	8.0+
Conductivity (micromhos)	04-17-74	T	19,200	--	23,400	24,300	23,100	26,300
	04-17-74	B	27,400	--	32,000	32,000	30,100	30,200
	05-21-74	T	34,600	34,900	34,700	35,000	35,400	36,400
	05-21-74	B	34,300	34,000	34,700	34,000	--	--
	07-09-74	T	29,600*	--	33,300	35,500	36,300	37,100+
	07-09-74	B	33,100*	--	35,200	34,600	33,100	30,600+
Salinity (g/1)	04-17-74	T	17.0	--	21.4	22.4	20.7	24.0
	04-17-74	B	26.2	--	31.7	31.6	29.1	28.4
	05-21-74	T	29.1	29.3	28.2	28.6	29.0	29.5
	05-21-74	B	29.8	30.2	30.3	30.2	--	--
	07-09-74	T	21.0*	--	22.9	25.5	25.8	27.8+
	07-09-74	B	24.8*	--	28.5	28.7	28.9	28.8+

T Top - Approximately 2 ft. below surface
B Bottom - Approximately 2 ft. above bottom

* Samples Collected 06-11-74
+ Samples Collected 06-14-74

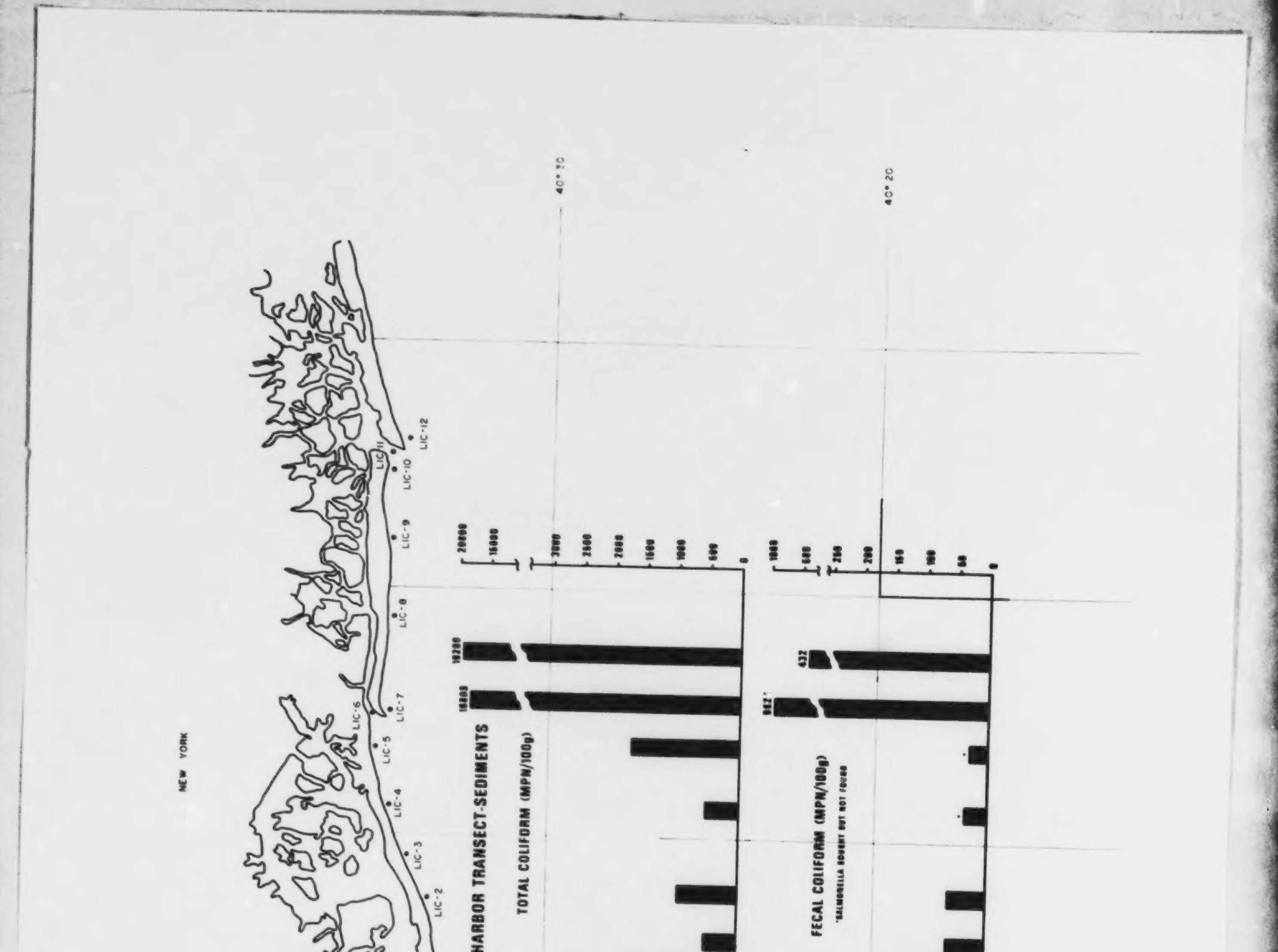
NEW YORK HARBOR TRANSECT-SEDIMENTS
TOTAL COLIFORM (MPN/100g)
FECAL COLIFORM (MPN/100g)
*SALMONELLA SOUGHT BUT NOT FOUND
20000
15000
3000
2500
2000
1500
1000
500
0
1000
500
250
200
150
100
50
0
2860
19200
18800
432
842*
284*
NEW YORK
UPPER BAY
LOWER BAY
SANDY HOOK BAY
LONG BRANCH
ASBURY PARK
NEW JERSEY
LIC-1
LIC-2
LIC-3
LIC-4
LIC-5
LIC-6
LIC-7
LIC-8
LIC-9
LIC-10
LIC-11
LIC-12
JC-01
JC-02
JC-03
JC-05
JC-08
JC-11
JC-14
JC-21
40° 30
40° 20

Figure 7

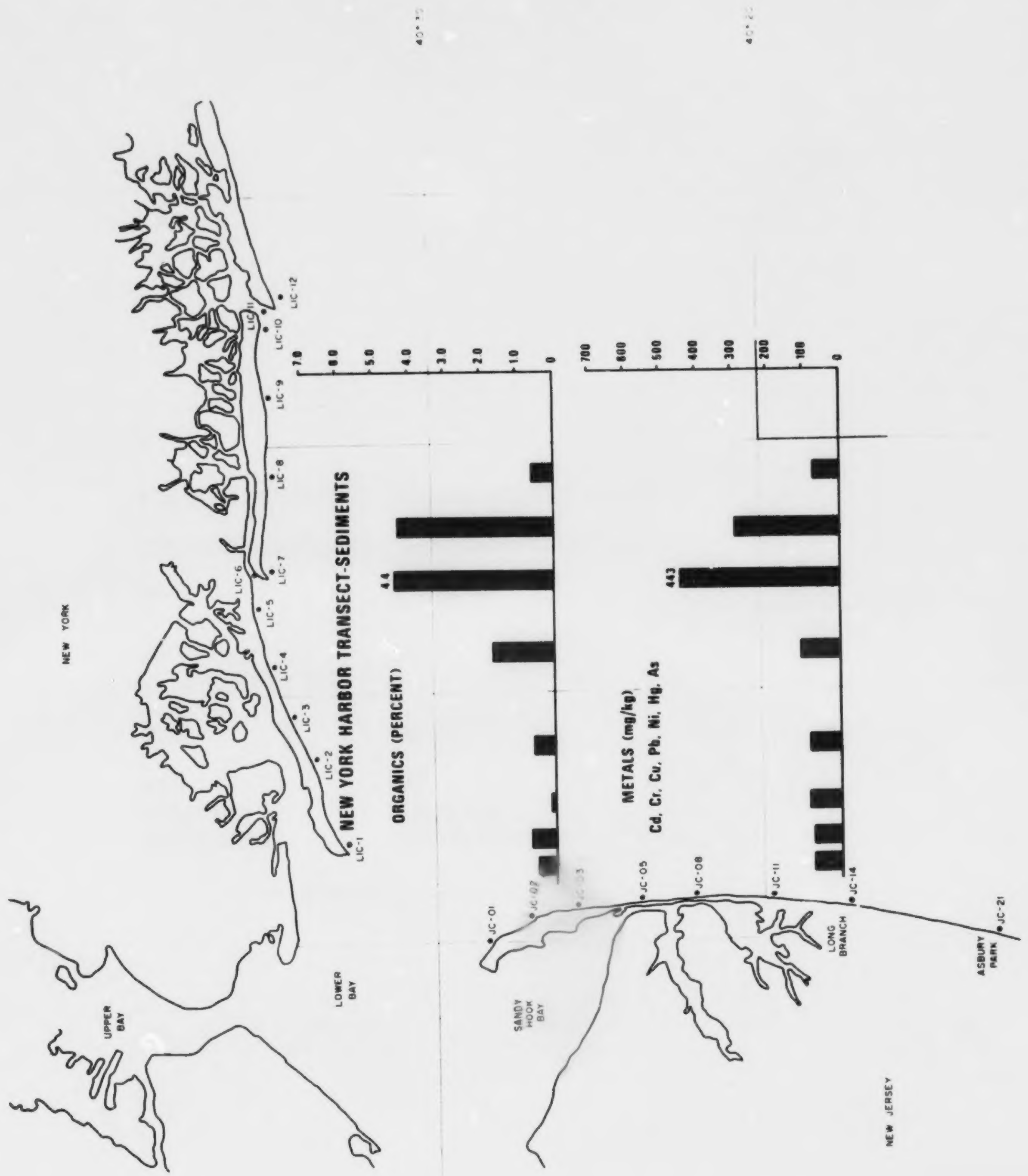
NEW YORK HARBOR TRANSECT-SEDIMENTS
ORGANICS (PERCENT)
70
60
50
40
30
20
10
0
44
METALS (mg/kg)
Cd, Cr, Cu, Pb, Ni, Hg, As
700
600
500
400
300
200
100
0
443
NEW YORK
UPPER BAY
LOWER BAY
SANDY HOOK BAY
LONG BRANCH
ASBURY PARK
NEW JERSEY
LIC-1
LIC-2
LIC-3
LIC-4
LIC-5
LIC-6
LIC-7
LIC-8
LIC-9
LIC-10
LIC-11
LIC-12
JC-01
JC-02
JC-03
JC-05
JC-08
JC-11
JC-14
JC-21

Figure 7a

TABLE 7

SUMMARY OF WATER-QUALITY DATA (BOTTOM SEDIMENTS) COLLECTED ALONG N.Y. HARBOR TRANSECT

Parameter	Date	Station Identification Number					
		NYB30	NYB31	NYB32	NYB33	NYB34	NYB35
Total Coliform	04-17-74	2,300	--	1,300	790	1,300	2,300
(MPN/100 g)	05-21-74	1,410	1,720	230	1,410	490	1,720
	07-09-74	7,900*	--	490	790	230	1,300+
Fecal Coliform	04-17-74	230	--	130	130	20	50
(MPN/100 g)	05-21-74	80	170	20	110	50	20
	07-09-74	460*	--	110	20	50	20+
Salmonella	04-17-74	--	--	--	--	Neg.	Neg.
(qualitative)	06-11-74	Neg.	--	--	--	--	--
Total Organics	04-17-74	3,490	--	2,580	7,190	27,700	60,900
(mg/kg)	05-21-74	4,630	6,660	3,080	6,260	11,500	32,500
	07-09-74	6,530*	--	1,290	5,380	11,400	38,200+
Cadmium	04-17-74	--	--	--	--	--	3
(mg/kg)	05-21-74	< 2	< 2	< 2	< 2	< 2	--
	07-09-74	< 3*	--	< 5	< 5	< 5	6+
Chromium	04-17-74	--	--	--	--	--	74
(mg/kg)	05-21-74	3	7	3	9	9	--
	07-09-74	4*	--	< 10	11	17	19+
Copper	04-17-74	--	--	--	--	--	82
(mg/kg)	05-21-74	< 6	< 6	< 6	8	42	--
	07-09-74	< 6*	--	3	7	13	140+
Lead	04-17-74	--	--	--	--	--	134
(mg/kg)	05-21-74	< 50	< 50	< 50	< 50	50	--
	07-09-74	< 50*	--	< 50	< 50	< 50	250+
Nickel	04-17-74	--	--	--	--	--	17
(mg/kg)	05-21-74	< 7	< 7	< 7	< 7	8	--
	07-09-74	10*	--	24	< 10	13	20+
Mercury	04-17-74	--	--	--	--	2.1	4.4
(mg/kg)	05-21-74	.2	.5	< .2	1.1	1.0	--
	07-09-74	< .1*	--	.06	.21	.31	1.7+
Arsenic	04-17-74	--	--	--	--	--	2.4
(mg/kg)	05-21-74	1.4	1.8	1.6	4.9	4.5	--
	07-09-74	3.6*	--	.8	1.8	2.4	12+

* Samples Collected 06-11-74
+ Samples Collected 06-14-74

Neg. - Sought but not detected

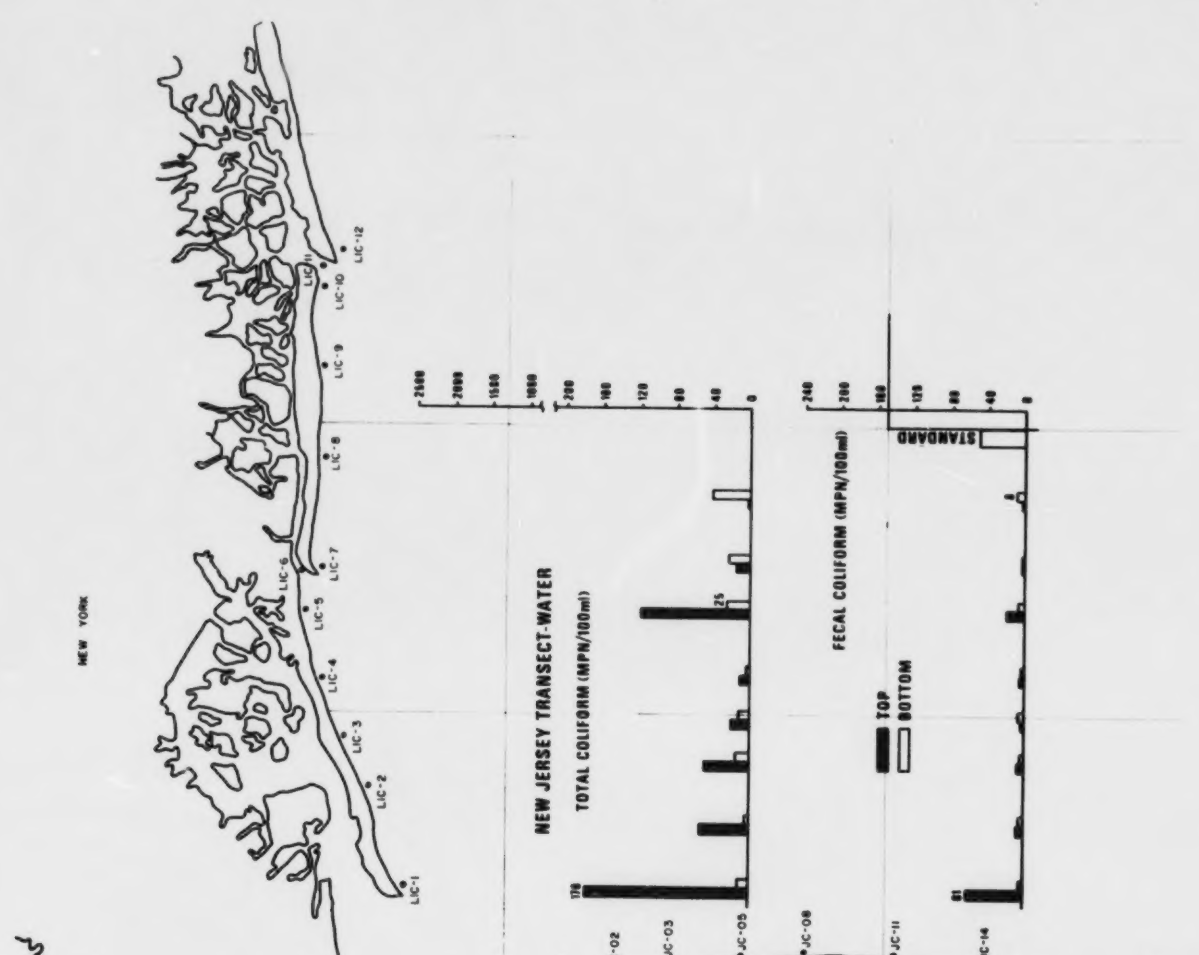
NEW YORK
UPPER BAY
LOWER BAY
SANDY HOOK BAY
NEW JERSEY
LONG BRANCH
ASBURY PARK
LIC-1
LIC-2
LIC-3
LIC-4
LIC-5
LIC-6
LIC-7
LIC-8
LIC-9
LIC-10
LIC-11
LIC-12
JC-01
JC-02
JC-03
JC-05
JC-08
JC-11
JC-14
JC-21
NEW JERSEY TRANSECT-WATER
TOTAL COLIFORM (MPN/100ml)
2500
2000
1500
1000
200
160
120
80
40
0
170
25
FECAL COLIFORM (MPN/100ml)
240
200
160
120
80
40
0
91
STANDARD
TOP
BOTTOM

Figure 8

TABLE 8

SUMMARY OF WATER-QUALITY DATA (WATER COLUMN) COLLECTED ALONG NEW JERSEY TRANSECT

Parameter	Date	Top/ Bottom	Station Identification Number							
			NYB20	NYB21	NYB22	NYB23	NYB24	NYB25	NYB26	NYB27
Total Coliform (MPN/100 ml)	04-18-74	T	460	17	70	79	46	3,480	79*	8
	04-18-74	B	5	< 2	13	< 2	2	9	490*	1,720
	05-16-74	T	2,400	33	22	< 2	2	< 2+	< 2	< 2$
	05-16-74	B	27	2	2	7	< 2	14+	2	< 2$
	06-14-74	T	5	79	79	49	11	230	17	< 2
	06-14-74	B	13	22	141	130	11	130	13	22
Fecal Coliform (MPN/100 ml)	04-18-74	T	230	5	5	5	5	175	2*	2
	04-18-74	B	< 2	< 2	< 2	< 2	< 2	2	2*	33
	05-16-74	T	490	7	7	< 2	< 2	< 2+	< 2	< 2$
	05-16-74	B	2	< 2	< 2	2	< 2	2+	< 2	< 2$
	06-14-74	T	< 2	< 2	5	2	5	22	2	< 2
	06-14-74	B	8	5	14	33	< 2	79	2	7
Water Temperature (°C)	04-18-74	T	7.9	7.6	7.8	7.8	7.8	7.8	7.2*	7.9
	04-18-74	B	5.9	6.1	7.8	7.1	6.9	7.1	6.9*	7.2
	05-16-74	T	14.1	9.9	9.9	10.2	10.8	11.7+	11.6	14.1$
	05-16-74	B	9.3	8.7	9.3	9.7	10.1	11.5+	10.7	--$
	06-14-74	T	18.2	17.9	18.1	17.7	17.5	17.1	16.9	17.9
	06-14-74	B	12.2	9.2	9.4	8.1	7.3	10.6	11.4	9.8
Dissolved Oxygen (mg/1)	04-18-74	T	8.7	7.3	7.6	6.4	11.2	12.0	8.0*	8.8
	04-18-74	B	4.5	8.2	9.1	6.4	9.2	7.2	9.5*	9.1
	05-16-74	T	11.0	7.6	9.2	9.0	9.8	4.5+	8.7	9.2$
	05-16-74	B	7.4	8.9	8.3	8.1	9.0	2.7+	8.0	6.7$
	06-14-74	T	9.8	10.4	11.4	11.2	10.0	9.3	9.2	10.0
	06-14-74	B	9.2	4.5	7.5	5.0	10.0	6.6	9.1	5.2
Conductivity (micromhos)	04-18-74	T	24,700	30,100	29,900	31,000	30,400	30,500	27,300*	29,100
	04-18-74	B	31,800	31,300	29,900	31,800	31,500	31,200	30,100*	30,700
	05-16-74	T	25,900	33,300	34,100	34,200	34,500	34,800+	34,900	36,900$
	05-16-74	B	32,800	33,200	33,900	34,100	34,300	34,900+	34,600	--$
	06-14-74	T	34,000	33,100	34,000	34,500	36,600	31,200	37,100	37,300
	06-14-74	B	33,600	31,800	31,700	31,100	30,700	32,600	33,200	32,000
Salinity (g/1)	04-18-74	T	22.4	28.2	27.7	29.0	28.4	28.4	25.5*	26.7
	04-18-74	B	31.4	30.9	27.8	30.3	30.2	29.8	28.6*	29.2
	05-16-74	T	20.2	29.8	30.3	30.3	30.0	29.8+	29.7	29.6$
	05-16-74	B	29.4	30.4	30.8	30.6	30.4	29.8+	30.3	--$
	06-14-74	T	24.8	24.5	25.1	25.6	27.3	28.1	28.2	27.9
	06-14-74	B	28.2	28.3	28.4	28.9	29.4	28.6	28.4	28.5

T Top - Approximately 2 ft. below surface
B Bottom - Approximately 2 ft. above bottom

* Samples Collected 04-17-74
+ Samples Collected 05-14-74
$ Samples Collected 05-21-74

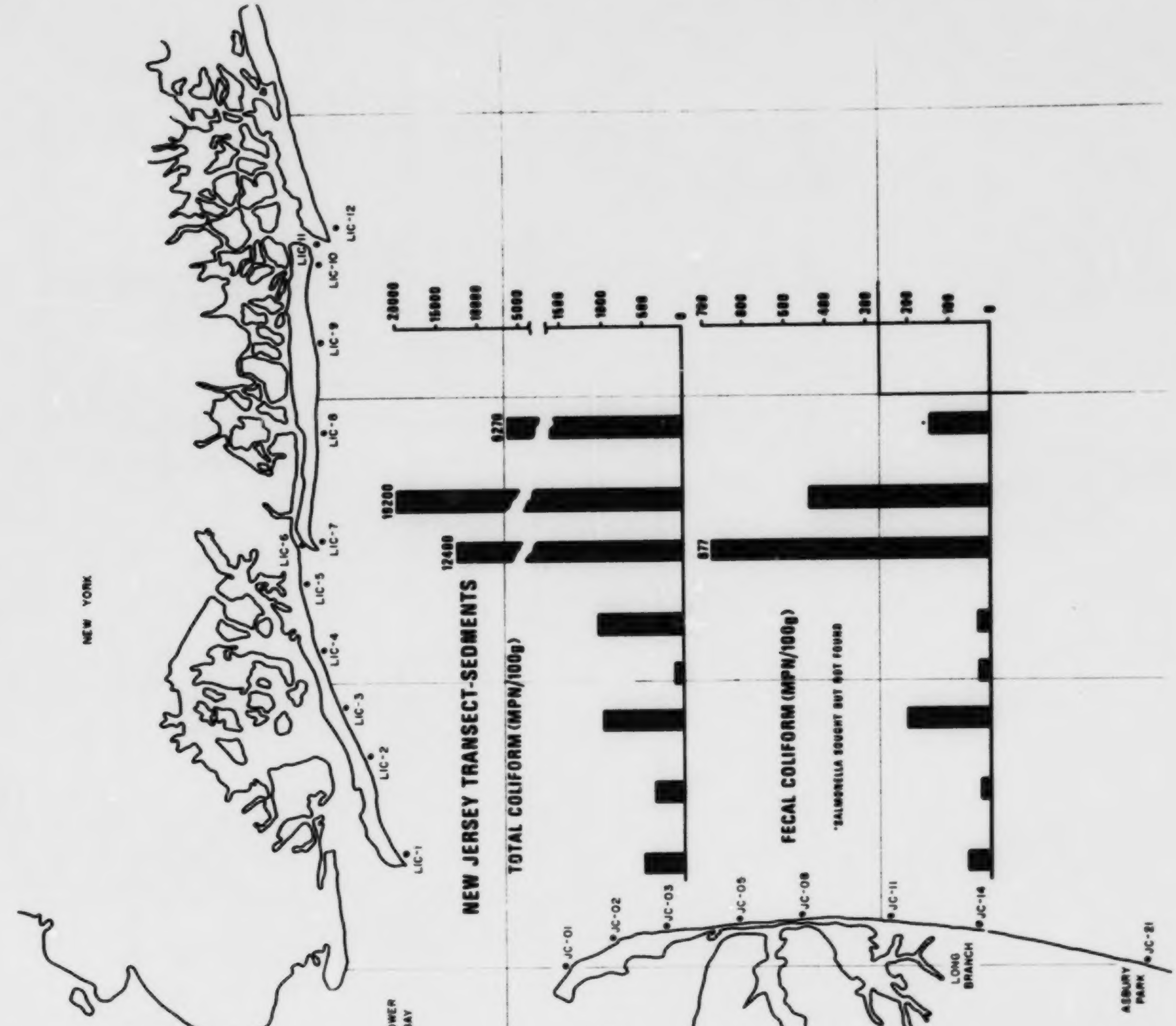
NEW JERSEY TRANSECT-SEDIMENTS
TOTAL COLIFORM (MPN/100g)
20000
15000
10000
5000
1500
1000
500
0
12400
19200
9270
FECAL COLIFORM (MPN/100g)
*SALMONELLA SOUGHT BUT NOT FOUND
700
600
500
400
300
200
100
0
877
NEW YORK
UPPER BAY
LOWER BAY
SANDY HOOK BAY
NEW JERSEY
LONG BRANCH
ASBURY PARK
LIC-1
LIC-2
LIC-3
LIC-4
LIC-5
LIC-6
LIC-7
LIC-8
LIC-9
LIC-10
LIC-11
LIC-12
JC-01
JC-02
JC-03
JC-05
JC-08
JC-11
JC-14
JC-21

Figure 9

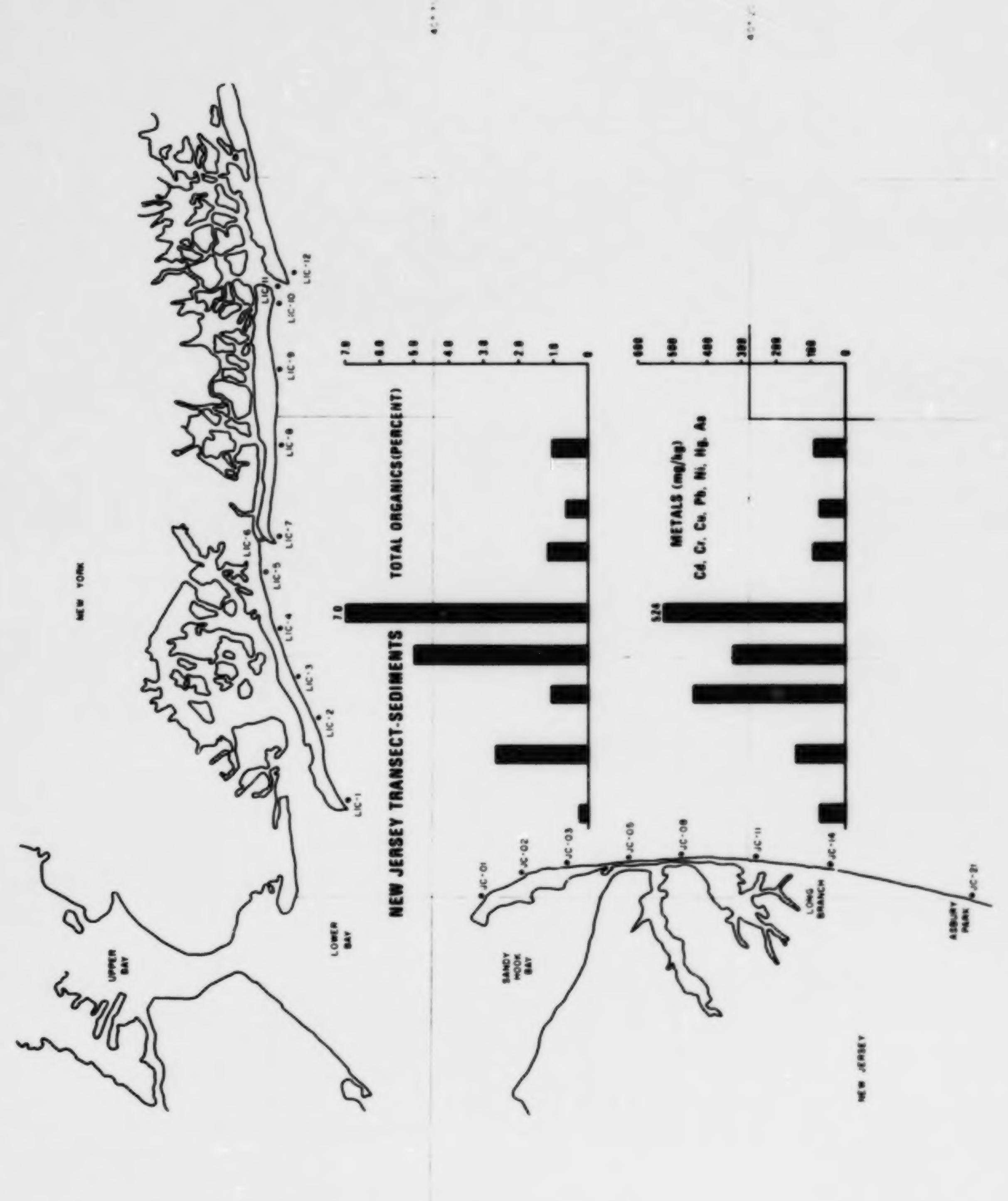
NEW YORK
UPPER BAY
LOWER BAY
SANDY HOOK BAY
LONG BRANCH
ASBURY PARK
NEW JERSEY
LIC-1
LIC-2
LIC-3
LIC-4
LIC-5
LIC-6
LIC-7
LIC-8
LIC-9
LIC-10
LIC-11
LIC-12
JC-01
JC-02
JC-03
JC-05
JC-08
JC-11
JC-14
JC-21
NEW JERSEY TRANSECT-SEDIMENTS
TOTAL ORGANICS (PERCENT)
7.0
6.0
5.0
4.0
3.0
2.0
1.0
0
7.0
METALS (mg/kg)
Cd, Cr, Cu, Pb, Ni, Hg, As
600
500
400
300
200
100
0
524

Figure 9a

TABLE 9

SUMMARY OF WATER-QUALITY DATA (BOTTOM SEDIMENTS) COLLECTED ALONG NEW JERSEY TRANSECT

Parameter	Date	Station Identification Number							
		NYB20	NYB21	NYB22	NYB23	NYB24	NYB25	NYB26	NYB27
Total Coliform	04-18-74	2,300	130	490	20	790	7,900	92,000*	24,000
(MPN/100 g)	05-16-74	230	490	1,090	220	1,300	7,000+	1,410	7,900$
	06-14-74	230	490	1,720	230	1,300	34,800	54,200	1,300
Fecal Coliform	04-18-74	330	< 20	110	< 20	< 20	80	490*	490
(MPN/100 g)	05-16-74	20	20	230	< 20	50	490+	50	130$
	06-14-74	< 20	20	330	50	20	7,900	3,300	50
Salmonella (qualitative)	04-18-74	--	--	--	--	--	--	--	Neg.
Total Organics	04-18-74	1,640	31,500	13,200	--	96,700	7,030	5,670*	7,670
(mg/kg)	05-16-74	2,840	24,000	12,300	72,600	73,800	13,000+	4,850	11,300$
	06-14-75	3,330	24,000	18,600	27,900	38,900	13,400	7,740	11,600
Cadmium	04-18-74	--	--	< 3	3	13	< 3	< 3*	--
(mg/kg)	05-16-74	< 2	< 2	< 2	--	--	--	--	< 2$
	06-14-74	< 3	< 3	< 3	< 3	< 3	< 3	< 3	< 3
Chromium	04-18-74	--	--	38	87	240	10	9*	--
(mg/kg)	05-16-74	< 3	33	19	--	--	--	--	10$
	06-14-74	5	40	31	59	41	10	8	14
Copper	04-18-74	--	--	47	100	260	5	5*	--
(mg/kg)	05-16-74	< 6	34	35	--	--	--	--	7$
	06-14-74	< 6	34	30	68	37	10	6	9
Lead	04-18-74	--	--	58	126	302	< 40	< 40*	--
(mg/kg)	05-16-74	< 50	< 50	< 50	--	--	--	--	< 50$
	06-14-74	< 50	58	52	118	70	< 50	< 50	< 50
Nickel	04-18-74	--	--	12	24	39	< 10	5*	--
(mg/kg)	05-16-74	< 7	10	13	--	--	--	--	< 7$
	06-14-74	< 10	14	13	25	21	< 10	< 10	< 10
Mercury	04-18-74	--	--	3.8	4.0	8.7	2.9	2.2*	--
(mg/kg)	05-16-74	< .2	.7	.4	--	--	--	--	.3$
	06-14-74	.1	.7	.35	1.0	.75	< .1	< .1	< .1
Arsenic	04-18-74	--	--	3.6	12.8	2.4	5.0	3.0*	--
(mg/kg)	05-16-74	2.8	4.7	3.4	--	--	--	--	3.7$
	06-14-74	4.1	4.7	2.6	14	11	13	--	3.3

* Samples Collected 04-17-74
+ Samples Collected 05-14-74
$ Samples Collected 05-21-74

Neg. - Sought but not detected

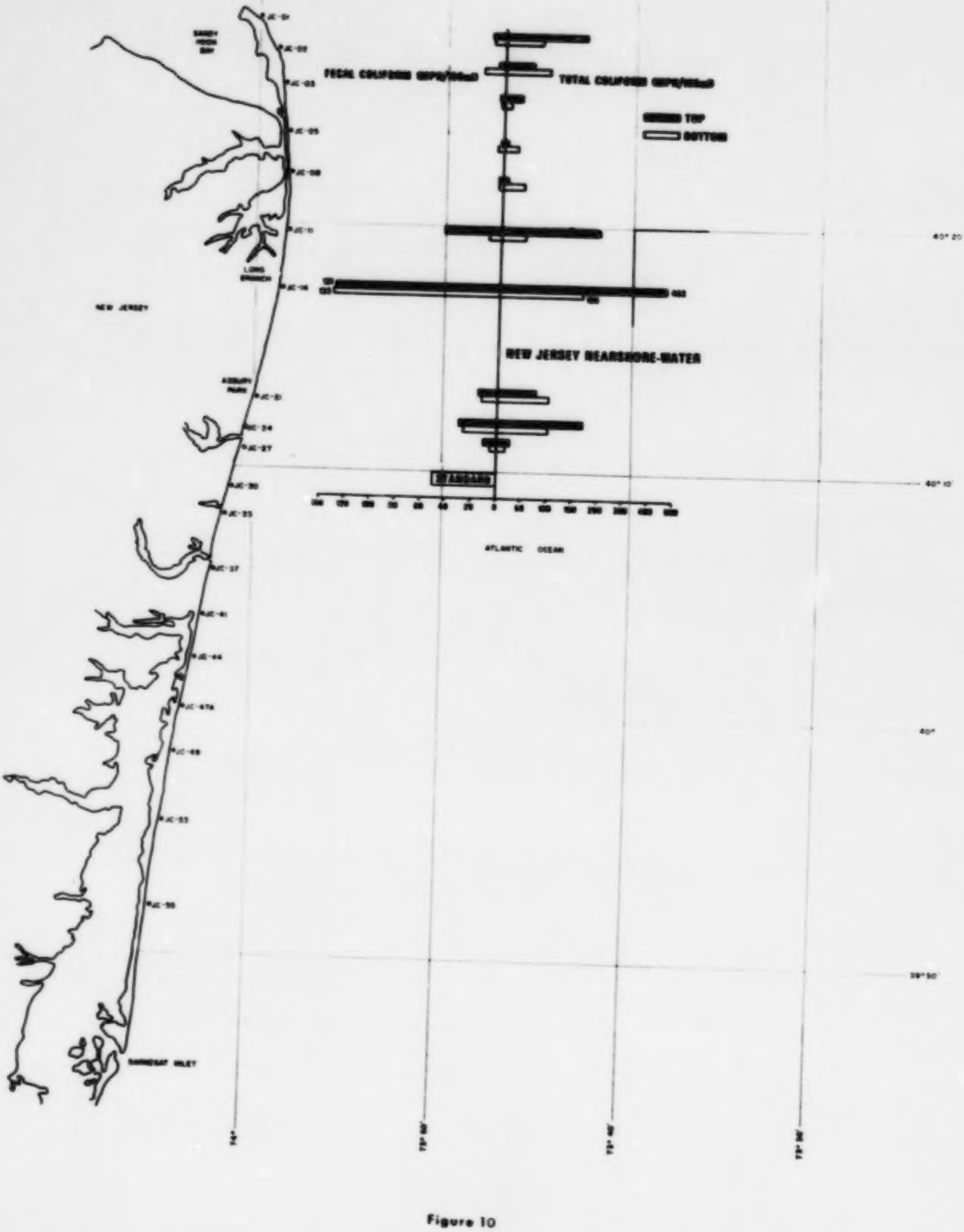
SANDY HOOK BAY
FECAL COLIFORM (MPN/100ml)
TOTAL COLIFORM (MPN/100ml)
TOP
BOTTOM
JC-01
JC-02
JC-03
JC-05
JC-08
JC-11
LONG BRANCH
JC-14
NEW JERSEY
NEW JERSEY NEARSHORE-WATER
ASBURY PARK
JC-21
JC-24
JC-27
STANDARD
JC-30
JC-33
0
ATLANTIC OCEAN
JC-37
JC-41
JC-44
JC-47A
JC-49
JC-53
JC-55
BARNEGAT INLET
40° 20'
40° 10'
40°
39° 50'
74°
73° 50'
73° 40'
73° 30'

Figure 10

TABLE 10

SUMMARY OF WATER-QUALITY DATA (WATER COLUMN) COLLECTED ALONG THE NEW JERSEY COAST
(by boat approximately 100 yards from beach)

Parameter	Date	Top/ Bottom	Station Identification Number									
			JC01	JC02	JC03	JC05	JC08	JC11	JC14	JC21	JC24	JC27
Total Coliform (MPN/100 ml)	04-06-74	T	141	70	23	13	11	9	22	17	22	2
	04-06-74	B	330	79	33	79	17	8	14	34	49	2
	04-30-74	T	130	120	49	23	70	33	490	330	1,300	278
	04-30-74	B	94	221	33	49	490	172	3,480	49	172	330
	07-10-74	T	230	23	23	2	2	> 24,000	9,200	79	172	33
	07-10-74	B	13	33	2	7	11	790	1,300	700	130	8
Fecal Coliform (MPN/100 ml)	04-06-74	T	14	13	2	< 2	2	2	5	2	2	< 2
	04-06-74	B	17	23	5	5	2	< 2	< 2	5	5	< 2
	04-30-74	T	17	9	8	2	8	2	130	49	109	49
	04-30-74	B	11	46	4	13	5	2	2,400	14	49	34
	07-10-74	T	5	2	8	< 2	< 2	> 24,000	3,480	33	130	11
	07-10-74	B	5	5	< 2	< 2	2	230	490	23	79	2
Water Temperature (°C)	04-06-74	T	16.1	15.7	16.9	16.6	16.3	14.9	14.8	15.1	14.7	15.5
	04-06-74	B	16.4	15.8	16.9	16.8	16.5	15.6	15.1	15.3	15.3	15.5
	04-30-74	T	14.4	11.1	10.5	11.1	9.7	9.7	8.4	10.0	9.5	9.8
	04-30-74	B	10.0	8.7	10.5	8.8	8.3	8.5	8.5	9.3	8.2	8.6
	07-10-74	T	21.9	21.6	21.8	21.4	21.2	21.7	22.3	20.7	21.0	21.6
	07-10-74	B	19.6	21.1	20.1	17.6	16.7	15.7	18.1	17.8	18.8	18.2
Dissolved Oxygen (mg/l)	04-06-74	T	7.0	7.7	7.6	7.8	8.2	7.0	8.2	9.3	7.0	8.3
	04-06-74	B	7.0	8.2	8.3	8.7	9.3	8.4	9.5	9.0	8.2	8.4
	04-30-74	T	12.4	12.9	12.0	12.0	11.3	11.3	11.3	10.8	10.1	9.5
	04-30-74	B	11.1	10.2	11.6	10.5	11.2	9.0	9.2	11.2	10.5	6.5
	07-10-74	T	10.0	9.4	10.2	10.5	8.5	9.2	10.2	8.6	7.9	8.6
	07-10-74	B	7.6	9.1	7.2	4.2	3.5	2.5	4.6	7.4	6.3	5.6
Conductivity (micromhos)	04-06-74	T	29,500	31,600	32,000	32,700	33,000	33,600	33,400	33,400	33,400	33,700
	04-06-74	B	29,200	31,700	32,000	32,100	32,600	33,800	33,900	33,400	33,200	33,700
	04-30-74	T	29,400	29,700	30,000	29,700	31,200	30,400	30,800	31,700	32,500	33,100
	04-30-74	B	30,400	31,600	30,200	31,700	32,800	30,500	32,600	32,100	32,900	33,400
	07-10-74	T	34,100	35,900	36,600	38,600	37,500	38,300	38,200	39,100	39,300	39,800
	07-10-74	B	36,100	36,400	36,800	37,300	36,800	36,500	38,300	38,100	38,500	38,200
Salinity (g/l)	04-06-74	T	22.3	24.2	24.1	24.7	25.2	26.4	26.1	26.1	26.3	26.4
	04-06-74	B	21.9	24.3	24.1	24.2	24.7	26.1	26.4	25.8	25.7	26.0
	04-30-74	T	23.3	25.2	25.9	25.4	27.7	27.0	26.8	27.9	29.5	29.1
	04-30-74	B	26.7	28.8	26.3	28.8	30.5	27.6	30.0	28.8	30.5	30.4
	07-10-74	T	23.0	24.5	24.9	26.6	25.9	26.2	25.9	27.6	27.3	27.6
	07-10-74	B	25.7	27.4	26.0	27.2	27.8	28.4	28.0	28.2	28.1	28.1

T Top - Approximately 2 ft. from surface
B Bottom - Approximately 2 ft. from bottom

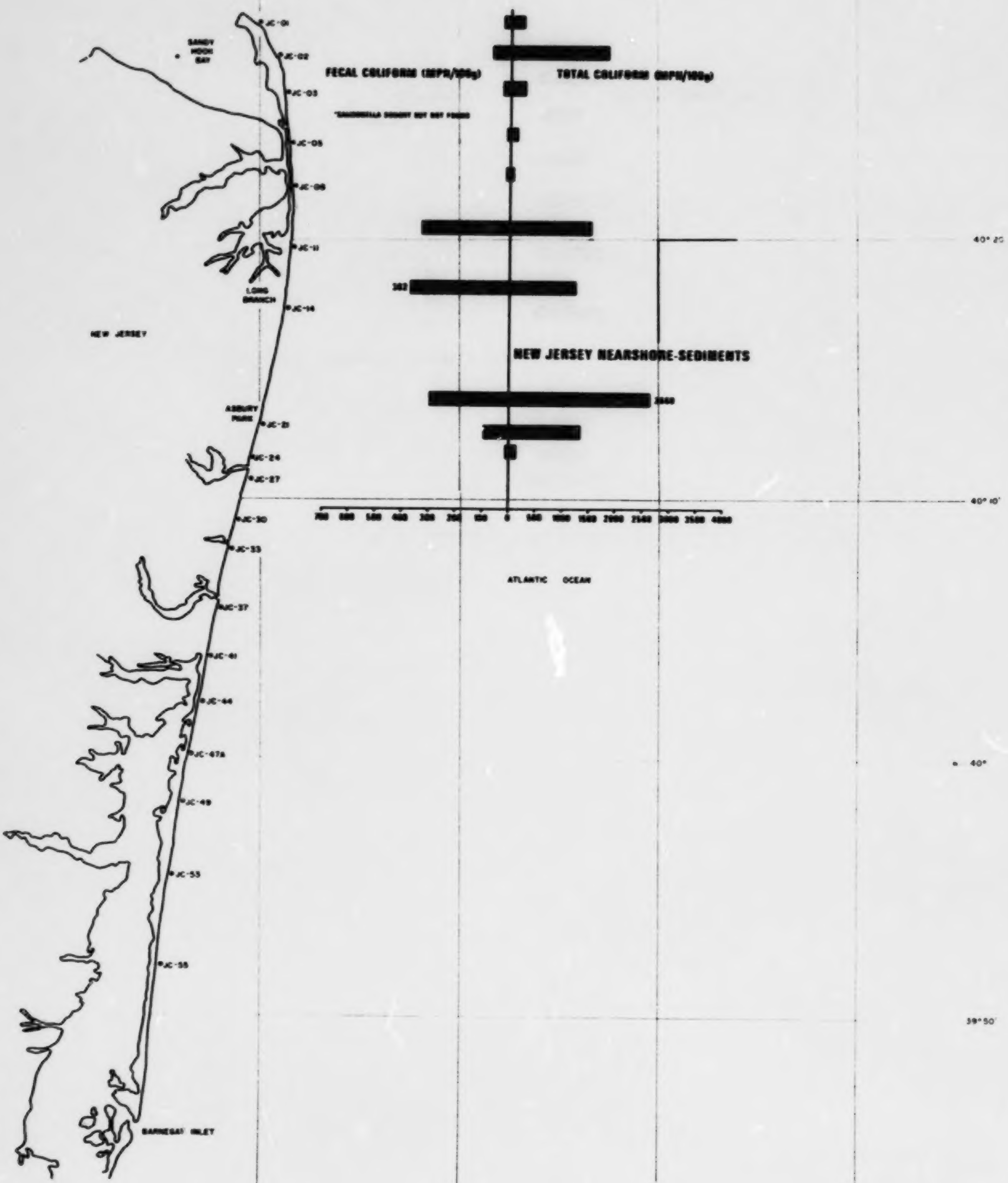
SANDY HOOK BAY
JC-01
JC-02
JC-03
JC-05
JC-08
JC-11
LONG BRANCH
JC-14
NEW JERSEY
ASBURY PARK
JC-21
JC-24
JC-27
JC-30
JC-33
JC-37
JC-41
JC-44
JC-47A
JC-49
JC-53
JC-55
BARNEGAT INLET
FECAL COLIFORM (MPN/100g)
TOTAL COLIFORM (MPN/100g)
*SALMONELLA SOUGHT BUT NOT FOUND
382
NEW JERSEY NEARSHORE-SEDIMENTS
700
600
500
400
300
200
100
0
500
1000
1500
2000
2500
3000
3500
4000
ATLANTIC OCEAN
40° 20'
40° 10'
40°
39° 50'
74°
73° 50'
73° 40'
73° 30'

Figure 11

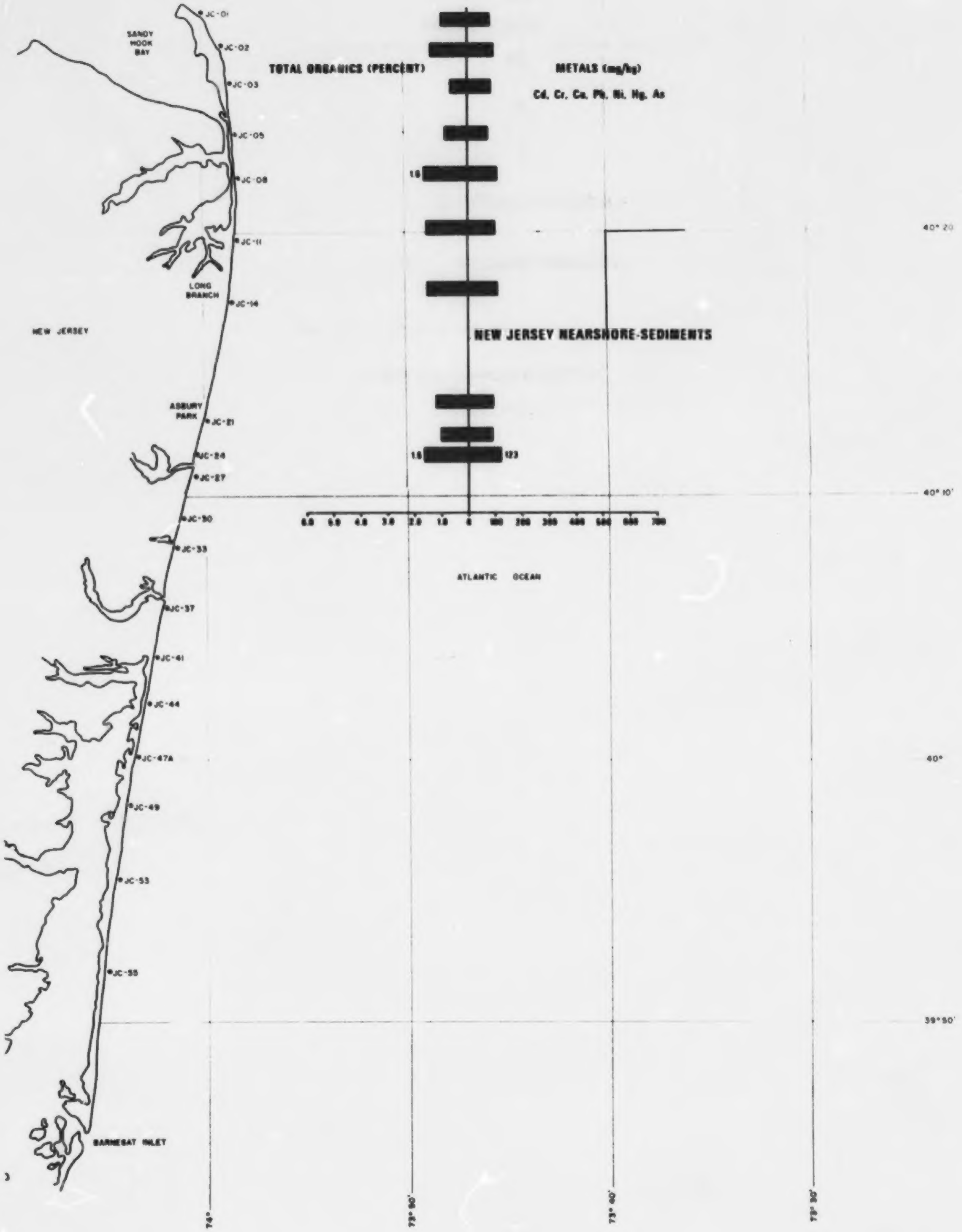
SANDY HOOK BAY
JC-01
JC-02
TOTAL ORGANICS (PERCENT)
METALS (mg/kg)
Cd, Cr, Cu, Pb, Ni, Hg, As
JC-03
JC-05
1.6
JC-08
40° 20'
JC-11
LONG BRANCH
JC-14
NEW JERSEY
NEW JERSEY NEARSHORE-SEDIMENTS
ASBURY PARK
JC-21
JC-24
1.6
123
JC-27
40° 10'
JC-30
6.0
5.0
4.0
3.0
2.0
1.0
0
100
200
300
400
500
600
700
JC-33
ATLANTIC OCEAN
JC-37
JC-41
JC-44
JC-47A
40°
JC-49
JC-53
JC-55
39° 50'
BARNEGAT INLET
74°
73° 50'
73° 40'
73° 30'

Figure 11a

TABLE 11

SUMMARY OF WATER-QUALITY DATA (BOTTOM SEDIMENTS) COLLECTED ALONG THE NEW JERSEY COAST
(by boat approximately 100 yards from beach)

Parameter	Date	Station Identification Number									
		JC01	JC02	JC03	JC05	JC08	JC11	JC14	JC21	JC24	JC27
Total Coliform (MPN/100 g)	04-06-74	1,720	17,200	130	230	130	50	790	3,300	1,300	230
	04-30-74	110	2,780	1,300	490	1,720	22,100	4,900	1,720	2,210	50
	07-10-74	80	130	130	20	80	3,300	490	3,300	790	230
Fecal Coliform (MPN/100 g)	04-06-74	230	1,300	< 20	< 20	20	< 20	110	330	50	< 20
	04-30-74	40	110	40	20	40	2,300	2,300	230	50	< 20
	07-10-74	< 2	< 2	20	< 2	< 2	790	220	330	330	< 2
Salmonella (qualitative)	04-06-74	--	--	--	--	--	Neg.	--	--	--	--
	04-30-74	--	--	--	Neg.	--	--	--	--	--	--
Total Organics (mg/kg)	04-06-74	11,000	17,700	7,130	8,500	15,300	15,800	18,900	13,300	11,300	14,200
	04-30-74	8,010	8,610	5,930	8,750	15,900	14,900	12,600	10,900	9,500	28,400
	07-10-74	7,460	6,350	6,440	7,400	11,120	16,340	9,880	9,460	11,020	9,680
Cadmium (mg/kg)	04-06-74	--	< 3	< 3	--	< 3	--	< 3	< 3	< 3	< 3
	04-30-74	< 5	< 5	--	< 5	--	< 5	--	< 5	--	< 5
	07-10-74	< 5	< 5	< 5	< 5	< 5	< 5	< 5	< 5	< 5	< 5
Chromium (mg/kg)	04-06-74	--	43	18	--	28	--	26	27	17	6
	04-30-74	9	15	--	13	--	27	--	16	--	51
	07-10-74	14	16	12	9	30	29	27	13	16	18
Copper (mg/kg)	04-06-74	--	14	< 6	--	< 6	--	< 6	< 6	< 6	< 6
	04-30-74	3	3	--	4	--	4	--	7	--	65
	07-10-74	5	< 3	< 3	3	16	5	< 3	< 3	7	4
Lead (mg/kg)	04-06-74	--	< 50	< 50	--	< 50	--	< 50	< 50	< 50	< 50
	04-30-74	40	< 40	--	< 40	--	< 40	--	< 40	--	< 40
	07-10-74	< 50	< 50	< 50	< 50	< 50	< 50	< 50	< 50	< 50	< 50
Nickel (mg/kg)	04-06-74	--	< 10	< 10	--	< 10	--	< 10	< 10	< 10	< 10
	04-30-74	< 10	< 10	--	< 10	--	< 10	--	< 10	--	34
	07-10-74	< 10	< 10	< 10	< 10	10	< 10	10	< 10	< 10	< 10
Mercury (mg/kg)	04-06-74	--	0.26	< 0.1	--	0.32	--	0.10	< 0.1	< 0.1	< 0.1
	04-30-74	0.1	< 0.1	--	0.2	--	< 0.1	--	0.8	--	0.1
	07-10-74	.42	.17	.95	.17	.08	.08	.06	.07	1.16	1.25
Arsenic (mg/kg)	04-06-74	--	11	2.1	--	9	--	12	3.7	3.0	3.8
	04-30-74	4.7	2.9	--	4.3	--	9.2	--	5.0	--	2.1
	07-10-74	1.9	2.1	1.4	1.7	3.0	3.6	2.6	2.7	4.7	2.8

Neg. - Sought but not detected

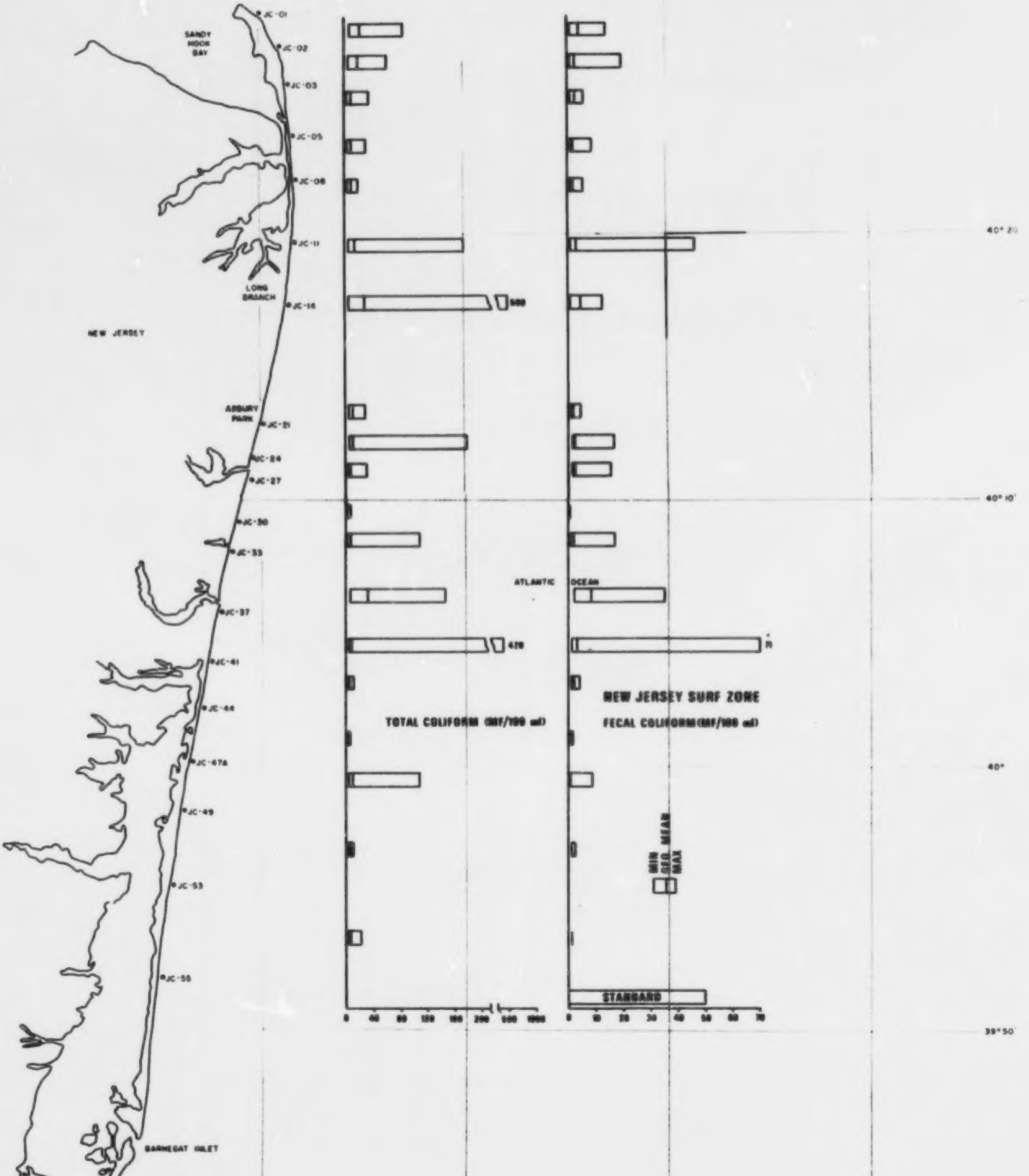
JC-01
SANDY HOOK BAY
JC-02
JC-03
JC-05
JC-08
JC-11
LONG BRANCH
JC-16
NEW JERSEY
ASBURY PARK
JC-21
JC-24
JC-27
JC-30
JC-33
JC-37
JC-41
JC-44
JC-47A
JC-49
JC-53
JC-55
BARNEGAT INLET
500
420
71
ATLANTIC OCEAN
NEW JERSEY SURF ZONE
TOTAL COLIFORM (MF/100 ml)
FECAL COLIFORM (MF/100 ml)
MIN
GEO. MEAN
MAX
STANDARD
0
40
80
120
160
200
500
1000
0
10
20
30
40
50
60
70
40° 20'
40° 10'
40°
39° 50'
74°
73° 50'
73° 40'
73° 30'

Figure 12

TABLE 12

SUMMARY OF BACTERIOLOGICAL DATA (WATER COLUMN) COLLECTED ALONG NEW JERSEY SHORELINE

(Samples Collected in Surf Zone)

Date	Station Identification Number									
	JC01A		JC02		JC03		JC05		JC08	
	TC	FC	TC	FC	TC	FC	TC	FC	TC	FC
4/8/74	12	1	4	1	10	1	12	<1	12	1
4/22/74	--	--	48	19	4	2	12	2	15	2
5/6/74	10	2	14	<1	8	5	8	<1	16	6
5/20/74	44	6	64	4	36	6	31	3	20	4
6/3/74	41	13	15	5	28	2	34	9	16	4
6/18/74	84	3	44	<1	--	--	11	4	10	2
7/1/74	6	<1	7	<1	3	--	1	1	1	<1
Samples to date	6	6	7	7	6	5	7	7	7	7
Geometric Mean	22	3	19	2	10	3	10	2	10	2
Max	84	13	64	19	36	6	34	9	20	6
Min	6	<1	4	<1	3	1	1	<1	1	<1

TC = Total Coliform (MF/100 ml)
FC = Fecal Coliform (MF/100 ml)
< = Less than

SUMMARY OF BACTERIOLOGICAL DATA (WATER COLUMN) COLLECTED ALONG NEW JERSEY SHORELINE

(Samples Collected in Surf Zone)

Date	Station Identification Number									
	JC11		JC14		JC21		JC24		JC27	
	TC	FC	TC	FC	TC	FC	TC	FC	TC	FC
4/8/74	6	1	18	9	--	--	4	2	8	4
4/22/74	7	<1	3	1	9	<1	2	1	<1	<1
5/6/74	176	47	23	5	31	5	3	1	4	1
5/20/74	12	5	500	13	9	1	2	1	2	2
6/3/74	10	1	17	6	14	3	16	3	10	2
6/18/74	18	3	64	8	23	3	168	17	19	16
7/1/74	5	<1	96	3	2	<1	5	<1	28	1
Samples to date	7	7	7	7	6	6	7	7	7	7
Geometric Mean	13	3	35	5	11	2	7	2	6	2
Max	176	47	500	13	31	5	168	17	28	16
Min	5	<1	3	1	2	<1	2	<1	<1	<1

TC = Total Coliform (MF/100 ml)
FC = Fecal Coliform (MF/100 ml)
< = Less than

SUMMARY OF BACTERIOLOGICAL DATA (WATER COLUMN) COLLECTED ALONG NEW JERSEY SHORELINE

(Samples Collected in Surf Zone)

Date	Station Identification Number									
	JC30		JC33		JC37		JC41		JC44	
	TC	FC	TC	FC	TC	FC	TC	FC	TC	FC
4/8/74	<2	<1	6	2	94	46	--	--	--	--
4/22/74	1	1	1	<1	10	3	--	--	--	--
5/6/74	2	<1	2	<1	15	6	6	1	10	2
5/20/74	<1	<1	<1	<1	8	2	7	7	<1	<1
6/3/74	8	<1	60	8	82	13	420	71	12	4
6/18/74	2	<1	112	17	148	32	4	<1	5	1
7/1/74	2	<1	28	1	35	2	1	<1	1	<1
Samples to date	7	7	7	7	7	7	5	5	5	5
Geometric Mean	2	<1	8	2	34	8	9	3	4	2
Max	8	1	112	17	148	46	420	71	12	4
Min	<1	<1	<1	<1	8	2	1	<1	<1	<1

TC = Total Coliform (MF/100 ml)
FC = Fecal Coliform (MF/100 ml)
< = Less than

TABLE 12 - Cont.

SUMMARY OF BACTERIOLOGICAL DATA (WATER COLUMN) COLLECTED ALONG NEW JERSEY SHORELINE

(Samples Collected in Surf Zone)

Date	Station Identification Number							
	JC47A		JC49		JC53		JC55	
	TC	FC	TC	FC	TC	FC	TC	FC
4/8/74	--	--	110	9	6	2	<2	<1
4/22/74	--	--	56	1	4	<1	<1	<1
5/6/74	1	1	2	1	8	2	23	<1
5/20/74	<1	<1	1	<1	<1	<1	1	1
6/3/74	3	2	6	1	5	<1	11	1
6/18/74	2	<1	2	<1	8	2	2	<1
7/1/74	1	<1	6	<1	2	<1	1	<1
Samples to date	5	5	7	7	7	7	7	7
Geometric Mean	1	1	7	1	4	1	3	<1
Max	3	2	110	9	8	2	23	1
Min	<1	<1	1	<1	<1	<1	<1	<1

TC = Total Coliform (MF/100 ml)
FC = Fecal Coliform (MF/100 ml)
< = Less than

DISPOSAL OF SEWAGE SLUDGE TO SEA;
UNITED KINGDOM EXPERIENCE AND PRACTICE

by

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BURNHAM-ON-CROUCH, ESSEX

Introduction

The disposal of sewage sludge to sea off the coasts of the United Kingdom is not a recent development. At least three large cities, London, Manchester and Glasgow, have used this method of sludge disposal for over fifty years. The scale of their operations has however increased considerably in recent years; in addition, many more local authorities are turning to the marine environment as a means of disposal for their sewage sludge. In 1971 Wood(1) stated that, in addition to London, Manchester and Glasgow, at least twelve other authorities disposed of sewage sludge to sea and the total quantity so disposed then amounted to 6947 x 10^3 wet tons per year. At the present time (May 1974) the number of other authorities dumping sewage sludge has risen to twenty-two and the total quantity to 8054 x 10^3 wet tons per year. The distribution in geographical terms is given in Table 1 (see Figure 1).

The reason for the upsurge in marine disposal of sewage sludge is twofold. Firstly, experience to date shows little accumulation of sewage sludge solids in any of the three main dumping areas (the Thames estuary, Liverpool Bay, and the outer Clyde estuary) and comparatively little effect on the benthic fauna and commercial fisheries in the areas of disposal. As a result the regulatory authorities continue to be favourably disposed towards sewage sludge disposal at sea and have allowed the increase in quantities.

The greater quantities dumped at sea arise primarily from increased pressure, from a variety of sources, against sludge disposal on land. First, transport costs have increased enormously, which renders land transport unattractive on economic grounds. Secondly, the Ministry of Agriculture, Fisheries and Food (MAFF) Agriculture Division has recently pressed strongly against the spreading of sewage sludges on land, because in the United Kingdom, in common with much of northern Europe, most sewage works treat a mixture of domestic and industrial wastes. This leads to the inclusion of appreciable quantities of metals in the resultant sludges, which may adversely affect the growth of certain crops and possibly lead to higher levels of these metals in foodstuffs grown on treated soils. These problems do not of course apply to sludges from rural communities.

Furthermore, regulatory authorities are reluctant to allow the solids removed in the course of sewage treatment to be discharged to rivers, and some authorities have recently exercised their powers in respect of drainage from several land-fill sites. Their objection against certain land-fill sites has been coupled with the saturation of several other major land-fill areas and a general shortage of alternative new sites. No place in the United Kingdom is more than about 70 miles from the sea, and it is perhaps not surprising that very many authorities have begun looking towards the sea as a means of solving their problems of sewage sludge disposal. Often sea disposal has an additional economic bonus in that cheap water transport down navigable rivers or estuaries can be used, and the more expensive road journeys can be reduced.

At present over 25% of the sludge produced for sewage treatment in the United Kingdom is discharged to sea, both from vessels and from long sea outfalls, and this quantity is likely to increase.

Policy and practice

The United Kingdom is a signatory to both the Oslo and London Dumping Conventions, and legislation which will allow ratification of these Conventions is currently passing through Parliament. The absence of formal controls does not however mean that UK dumping practice in either deep or shallow waters is, or has been, haphazard. For at least eight years control of disposal of materials to sea from the United Kingdom has been carried out under a voluntary scheme. At first this was operated by the then Board of Trade in consultation with the Fisheries Department of MAFF, but for the last five years or so MAFF has administrated the scheme in consultation with other interested parties such as the Department of Trade and Industry and the Department of the Environment.

Under this scheme any party wishing to dump waste at sea must apply for a licence and supply details of the waste. The details to be provided, usually on a standard application form, include mode of arising of the waste, its quantity and composition, and the proposed method and site of disposal. The administrative and scientific aspects of the scheme have been improved progressively and although in the past local authorities, and hence most sewage sludges, were not included in the scheme they are now.

When the new Dumping Bill receives the Royal Assent and becomes law, the voluntary scheme will become statutory and no dumping will be permitted without prior approval. Existing approvals will be given the status of licences and all new applications approved will receive formal licences.

In most cases, licences will be subject to conditions which will be reviewed at yearly intervals, and no guarantee of renewal of a licence will be given. Inspections will be conducted and a licence fee will be charged. Although it is not intended to make a profit the fee charged will cover all administrative, investigational and monitoring costs. The charges are unlikely to affect applications related to sewage sludge, but it is believed that they will deter the small-scale industrial applicant who in the past may simply have regarded marine disposal as a cheap and ready means of disposing of his wastes, without due consideration of alternative methods.

When application is received by MAFF for an approval to dump a waste at sea some applicants suggest a dumping site, but others leave this open for discussion with and guidance from MAFF. In either case before a site is approved many factors have to be taken into account. Some of these are related specifically to the composition and characteristics of the waste and for the purposes of this paper are considered later.

Other important factors are also considered and apply to any waste. Amongst the first of these are the navigational and operational aspects of the dumping area. The shipping lanes around the United Kingdom are among the busiest in the world and it is essential that interference with normal commercial shipping operations, as for example a dumping vessel depositing waste in the path of oncoming vessels, is avoided. This is an important practical consideration; thus one of the main reasons for the alteration of the agreed site for disposal of the sewage sludge from London was due to the original site being the main shipping channel into and out of the Port of London. It is also necessary to ensure that any accumulation of solids does not affect navigational channels.

There are extensive deposits of sand and gravel around the United Kingdom, and increasing quantities are being won and used in the construction industry in England and in continental Europe. These operations must be safeguarded, and care must be taken to avoid impairing the operation of dredgers and to ensure that the deposits which are being extracted, or which might be extracted in the future, are not fouled or contaminated in a way which would render the material unsuitable for construction purposes.

Interference with amenities must also be considered. With the increasing use of the sea for various forms of recreation, dumping grounds are sited sufficiently far from land to ensure that no unacceptable discoloration of water or offensive floating or suspended matter is detectable by the public. This includes the possibility of material being carried to the coast by wave action or local current systems, and is especially relevant in areas of high recreation value which might be threatened by dumping of raw or partially-treated sewage sludge. Experience has shown that in respect of sewage sludge dumping there is no evidence of increased number of faecal micro-organisms in bathing beach waters, and this has been confirmed by tracer tests(2); the effects of local coastal discharges from pipelines are far more important in this respect.

In many areas, the most important consideration is the possible effect on marine resources. The coastal waters around the United Kingdom are extremely productive and fishing activity is very heavy in many areas. In selecting a suitable area for dumping it is essential to ensure that interference with fishing activity is minimal. Although it is unlikely that sewage sludge will affect the physical operation of fishing gear by obstructing trawling, potting, etc., it is aesthetically undesirable that fishing should be carried out in an area of sewage sludge disposal. Due to the high numbers of faecal micro-organisms present, sewage sludge should not be dumped on or near exploited molluscan shellfish beds - in practice this is not a great problem in UK waters but it may be in other parts of the world.

Generally, the direct acute toxicity of sewage sludges is low, at least to commercially-sized marine fish or shellfish, but sub-lethal effects may be important. That part of the ecosystem on which commercial species depend for food must also be protected. The spawning grounds of species such as herring, sand eels and rays, whose eggs require a firm substrate on which to adhere and develop, are especially vulnerable. Such species are threatened not only by the smothering of deposited eggs but by the permanent or temporary loss of suitable substrate for subsequent spawnings. Fortunately, the major spawning grounds around the United Kingdom for such species are well known and can be safeguarded.

Nursery grounds of flatfish such as soles and plaice may also be harmed by the dumping of sewage sludge, because they often occur close inshore and near to large estuaries where many of our major towns are situated. Fortunately, these too are reasonably well established, following intensive inshore investigations by staff of the Ministry's Fisheries Laboratory at Lowestoft.

It is often stated that the solution to pollution is not simply a question of dilution. In general, however, marine resources can be protected from the polluting effects of sewage sludge, provided it is adequately dispersed either during or after disposal, and provided it does not subsequently reaccumulate in sludge banks or hollows in the sea bed. Lee and Ramster(3) have reviewed the various hydrographic techniques which have been used to measure or predict this in the context of pollution problems in the open sea.

The direction and extent to which a waste is transported after dumping is largely determined by the residual current flow. This can be measured through the water column by deploying moored recording current meters and long-term tracer experiments, often supplemented by a mathematical model, from which the likely movement of discharged materials can be satisfactorily predicted.

To predict the movement of sewage sludge, the rate of settlement of the solid particles is often important. Settlement will depend largely upon the density and particle size and can vary considerably with different sludges, due to the extent of dewatering and addition of flocculating aids. For example, the settlement rate of sewage sludge dumped from Manchester in Liverpool Bay was described as "low"(4), whereas MAFF scientists observing the release of sewage sludge in the outer Thames estuary reported settlement rates to be "quite rapid". Dispersal before settlement may be increased by the selection of a deep-water site with suitable physical oceanographic characteristics. Settlement may, however, be inhibited by stratification of the water column which may result in the temporary holding of sludge particles in the layer of discontinuity.

Once the sludge particles approach the sea bed their behaviour and fate is determined by their size and density, the bottom currents and the nature of the natural sea bed in the area of deposition. The currents at the sea bed may differ markedly from those at the surface, both in strength and direction, and the currents should therefore be measured at representative depths.

An inspection of the sediment types in the proposed dumping area and expected area of deposition can sometimes provide a general indication of the bottom currents. This can be achieved directly by divers, and by grab or dredge techniques, or indirectly by the use of echo sounders or other scanning devices. Fine sediments usually indicate low currents which would allow settlement, whereas sands and gravels generally indicate higher currents. These are only guidelines and several exceptions have been noted; for example, in an area of china clay waste disposal the natural sea bed was rocks and stones, but current velocities were low, the china clay settled out rapidly, and was retained in spite of winter storms(5).

Crickmore and Kiff(6) suggested that sludge particles are more readily absorbed on smooth mud than on rocks, where agitation associated with ripple movements discourages the retention of settled particles. The scale of the dumping operation is also important in this respect, since even strong currents may not be able to keep an area clear if the rate of deposition exceeds the carrying-capacity of the current.

The advent of the use of radioactive tracers has greatly increased the hydrographers' capacity to predict the transport of sewage sludge solids, both in the water and on the sea bed. By careful selection of the labelling isotope, initial dispersal and subsequent movement can be followed for several months. This technique has been used in connection with the studies made by MAFF and other scientists in both the Thames estuary (London sludge) and Liverpool Bay (Manchester sludge).

A final factor which is important in the United Kingdom, because of the considerable dumping of sludges derived from a heavily industrialized, densely populated island, is the possible interaction and synergism of toxic materials in the sewage sludge. The presence of existing discharges and of other dumping activities within a reasonable distance of the proposed dumping site are always taken into account.

Once the disposal site has been selected, for a sludge to be dumped in significant quantity, monitoring operations are instituted before any dumping takes place and are continued at intervals as necessary after dumping has commenced. At the present time the major emphasis is placed upon studies of the particle size composition of the sediment material, and the chemical composition of the sediments, animals and fish species, including those of non-commercial importance. The sediments are examined for their organic content and the presence of persistent substances such as metals, organochlorine pesticides and PCBs, where they are known to be present in the sewage sludge. The benthic animals and fish are also examined for the same persistent materials. Wherever possible, these chemical and physical studies are accompanied by short ecological surveys using grabs, dredges and Agassiz trawls. In spite of the fact that benthic populations can change very markedly, both in number and character, due to natural causes, these studies are considered essential, because they give indications to community structure and might provide warnings of changes that the chemist may not detect until later.

Experience

On the whole, British experience with sewage sludge disposal at sea has been satisfactory and the practice is generally viewed favourably. There have been a number of minor problems, usually caused by short dumping, e.g. where the dumping vessel discharged its cargo in the wrong area over a string of crab pots. However, although most applications are viewed favourably they do not automatically receive approval and some are refused. In common with the practice adopted for industrial wastes, each application is carefully considered on its merits.

The first consideration is the method of disposal, and the selection of a suitable disposal site in relation to fisheries, amenities and the ultimate fate of the sewage sludge solids. The general pattern of operation for the major authorities using sea disposal of the sewage sludge is to use hopper-type vessels, generally self-propelled but in a few cases of the dumb-barge type. These vessels discharge their cargo rapidly through bottom-opening doors; for instance, the London sludge vessels can discharge their entire cargo in about 5 minutes. Usually the vessel discharges under way. Many of the smaller authorities employ disposal contractors, and in general these operators use small coastal tankers with pumps discharging the sewage directly into the wake of the vessel.

It is estimated that the latter procedure provides a minimal initial dilution of around 1000, in contrast to the hopper-type operation which provides dilution of around 200-fold(4). It has been shown that subsequent dilution of 10-20-fold occurs(2) within an hour of disposal but that further dilution and dispersal is dependent on local tidal and climatic conditions. In general the aim for sewage sludges is to ensure maximum dispersal so as to avoid local build-up of sludge particles, de-oxygenation or hyper-nutrification.

Most disposal sites are situated outside the territorial limit of 3 miles but generally not more than 20 miles off. In no case have complaints been made of sewage sludges dumped at sea causing nuisance on beaches; however, in one instance an application to dump sewage sludge at sea was refused because of the risk that sludge would reach local beaches. This was due to the fact that the sludge in question had been dewatered and filter-pressed and was in pieces ranging from small particles up to quite large conglomerates which were either positively buoyant or only just negatively buoyant and liable to be moved on the surface by onshore winds. The composition of the sewage sludge is also taken into account. Both the Oslo and London Dumping Conventions require prohibition or limitation of the amounts of certain materials which may be dumped, and some of these may be present in sewage sludges.

The typical range of concentrations of various constituents of sewage sludges in the UK is given in Tables 2 and 3. The composition of a sewage sludge varies according to the nature of the crude sewage entering the treatment works and the degree of treatment. In the UK, raw primary sludge contains 3-8% dry solids, of which up to 20% might be grease; secondary sludge usually has a lower dry solids content of 1-3%, but also contains less grease and has a lower oxygen demand. Both primary and secondary sludges are dumped at sea, and the data in Tables 2 and 3 include both types.

From Table 2 it will be seen that the solids content of these wastes ranged between 1.3% and 60% (mean 8.4). Of these solids about 50-80% was organic matter, the remainder being fine grit, sand and other inorganic material. As stated earlier, quite a high proportion of the organic matter is grease or oil, typical contents falling in the range of 10-20% of the total solids. Generally, the majority of the oil is of mixed vegetable and animal origin, but occasionally quite high contents of mineral oil are encountered. It was on this count that applications have been refused, for it was found that the mineral oil content may exceed 3100 mg/kg wet weight; such wastes contravene legislation designed to control the discharge of oil at sea as required by the IMCO Convention. This legislation limits the discharge of oil or oily waste of any description, if it contains oil in excess of 100 ppm or if it is discharged at a rate in excess of 60 litres/mile. Most sewage sludges from large cities exceed this level and special dispensations are usually allowed, but in some cases the oil content is considered too high. The precise source of the mineral oil cannot always be traced but repeated sampling suggests it is likely to be of industrial origin. In this context it is believed that these difficulties arise as a result of the disposal by motorists of waste engine oil, much of which currently finds its way into the municipal sewer.

Only comparatively recently has it been realized that associated with the organic matter in sewage sludges there is an appreciable content of a wide range of metals and other persistent substances. The presence of these substances must be considered, since even if the sludge does not directly affect marine resources, the quality of benthic animals, fish or shellfish may be so affected as to render them unfit for human consumption. For example, mercury and cadmium - both of which, although permitted as trace quantities in other wastes, are specifically banned under the two Dumping Conventions - have been found to occur in concentrations of up to 20.4 mg/kg and 6.3 mg/kg respectively in UK sludges. Fortunately the normal or average content is much lower, i.e. 0.35 mg/kg for mercury and 1.1 mg/kg for cadmium. Other metals, such as lead, copper and zinc, must be controlled under the London Convention, and sewage sludges in the United Kingdom can contain considerable quantities; the maxima so far noted were 2130 mg/kg zinc, 171 mg/kg lead and 526 mg/kg copper, on a wet weight basis.

Investigations are in progress, both in the field and in the laboratory, in an attempt to determine whether or not all the metals in sewage sludges can enter the food chain and result in unacceptably high levels in edible fish and shellfish. From monitoring studies in sewage sludges disposal areas it has been shown that these substances are present in the sediments and in benthic organisms and in fish and shellfish which take them as food. In both the Clyde(8) and Thames(7) estuaries, high values of copper, lead and zinc were found in areas of high organic content sediments. There was a high degree of correlation between the zinc and copper and organic matter in the sediments in Liverpool Bay(10) but the zinc:copper ratio (9:1) was different from that in the sludge (2:1) or in harbour dredgings (6:1)(11). There was no clear correlation between the amounts of these metals in the sediment and in benthic molluscs, which suggests that the ecological relationships are very complex. Some of the analyses were carried out on whole animals and it is probable that the reported "tissue" levels refer, at least in part, to the metal associated with sediments contained in the gut of the animals. This metal may not be in a form available to the animal.

For some metals, particularly mercury, the evidence for their absorption is very strong. In both the Thames estuary and Liverpool Bay the mercury content of fish muscle is as high as that found in an area polluted by a chlor-alkali discharge (range of means 0.45-0.55 mg/kg; range of maxima 1.5-2.5 mg/kg). As is indicated in Table 3, sewage sludges also contain persistent organic substances such as organochlorine pesticides and PCBs. The concentrations found in sewage sludges from the UK cover a considerable range (e.g. for DDT between 0.01 mg/kg and 0.5 mg/kg), whereas for PCB the range was 0.1 mg/kg to 8.6 mg/kg. From our investigations, the average values for UK sludges appear to be about 0.2 mg/kg for DDT and 1.2 mg/kg for PCB. The source of these normal levels is obscure but in all probability is linked with levels in food and domestic sewage. Where abnormally high values occur they can usually be ascribed to loss from a specific industrial process; for example, 8.6 mg/kg PCB found in one sewage sludge was traced to losses of transmission fluids from railway locomotives(12). Similarly, certain sewage sludges from the Yorkshire woollen-processing area have been found to contain higher than average amounts of dieldrin, which was once used as a sheep dip and is still used for mothproofing(13).

Where urban sewage has a large industrial component, chlorinated solvents such as carbon tetrachloride and trichlorethylene may be found in sludges at concentrations of up to 20 mg/l on a dry weight basis(14).

Although the metal, organochlorine pesticide or PCB content of sewage sludge from UK sources does not normally give cause for concern, warnings have been given to dischargers that continuation of approval in the long term will be dependent upon significant reductions in the amount of some substances, such as cadmium, mercury or PCB. Frequently the discharger is unaware of the composition of his sewage sludge, or that it is exceptional, until discussions related to sewage composition are started. Usually, however, the principal source or sources of the pollutant are found and can be controlled.

It is always important to ensure that the requirements related to sludge quality take into account the input of similar substances from other sources. For example, in Liverpool Bay dredging spoil contributes a daily load of 3270 kg zinc, and sewage sludge adds a further 650 kg(1). Other sources such as input from pipelines and rivers may also be important, and aerial input can be considerable. For example, the annual rates of input of lead and copper respectively to the North Sea by aerial deposition have been estimated to be 15 000 tons and 13 000 tons respectively(15).

Where large amounts of organic matter and nutrients are dumped or discharged to the sea there is a danger that de-oxygenation or eutrophication might occur. In the United Kingdom, de-oxygenation has not proved to be a problem in sewage sludge disposal areas, largely because there are comparatively large tides which ensure considerable dispersion of the waste, and re-aeration of sea water. Tidal ranges (in metres) for the east coast of England and Scotland are shown in diagrammatic form in Figure 2. Clearly the currents associated with such tides are considerable.

The input of nutrients in sewage sludge is small by comparison with that from other sources; thus, in the North Sea only 7% of the nitrogen entering the North Sea can be attributed to sewage discharges of all types, and there is little evidence of eutrophication problems. It should however be stated that in at least one north European area (Oslofjord) the occurrence of a bloom of toxic algae and the incidence of paralytic shellfish poison toxin has been associated with the discharge of sewage effluents. In Liverpool Bay there is an annual bloom of the colonial flagellate Phaeocystis which can be toxic to young shellfish but this has not been linked with sewage sludge dumping(16).

The three major sewage sludge disposal operations around the United Kingdom, namely those from London, Manchester and Glasgow, have been thoroughly studied and this paper would not be complete without a brief summary of the findings, most of which have been published (London(9), Manchester(2) and (17) and Glasgow(18)).

The sewage sludge from London is dumped in the outer Thames estuary in a water depth of about 60 ft. In 1967 the area of disposal was altered from the Black Deep to the Barrow Deep, and both Deeps have been intensively studied since 1970. These studies have included dye releases, releases of radioactively-labelled sludge, current measurements and numerous surveys in which samples of sediment and animals have been collected for examination in the laboratory. The area is characterized by relatively deep channels separated by sandbanks, some of which can be exposed on extreme spring tides. The tidal currents in the present dumping zone are generally of the order of 2 knots but speeds greater than this have been observed.

In the first survey, which was conducted in 1970(9), no great accumulation of organic matter was found in either the old or new dumping zone, although there was an appreciable number of tomato seeds in the sediments in the Black Deep disposal zone. The main accumulation of organic matter was found inshore near the mouth of the Thames estuary. This organic matter was associated with elevated metal levels but the ratio of the metal was not the same as that in the sludge. Subsequent investigations using a radioactive tracer technique suggested that the high organic levels found at the mouth of the estuary were of estuarine origin rather than from the sewage sludge.

Owing to the fast currents and the high suspended matter load in the water of the Thames estuary, the numbers of benthic animals, particularly of bivalve molluscs, was not expected to be high and the surveys confirmed this. The distribution of bivalve molluscs did not appear to be linked to the sewage sludge disposal area, although it may be significant that most of the observed species were deposit feeders. The total numbers of polychaetes present appeared possibly to be related to the sludge disposal area, at least that of earlier years, but species diversity was also greatest in the area of greatest numbers.

Sludge from the city of Glasgow in Scotland has also been dumped to sea for almost seventy years. The current quantity dumped in the outer Clyde estuary is about 1 million tons per year. The characteristics of the disposal area are rather different from those in the Thames estuary; the shore is little more than 3 miles away on three sides but the water is deeper, about 300 ft. Current velocities are lower, generally < 0.5 knots. Releases of seabed drifters(18) have shown that light material such as sewage sludge would however be spread over a wide area relatively rapidly.

As might perhaps be expected, in view of the conditions in the disposal area, there is some evidence of an accumulation of organic matter in the sediments. However, the concentrations found were not unduly high - the highest being only 8%, i.e. about eight times the lowest observed in the survey area. Similarly, the levels of metals present in the dumping zone were in excess of those found further away, and the highest values were coincident with those for organic matter. Concentrations of up to 208 mg/kg of copper, 320 mg/kg of lead and 488 mg/kg of zinc were observed in the centre of accumulation. There was some evidence of a change in species composition which may be associated with the greater concentration of organic matter in the dumping area, since the change is from a basically molluscan/echinoderm community to a polychaete community in the organic-rich zone.

Analyses of mussels from selected sites(8) in the immediate dumping area and from a control area some distance away suggest that the animals in the disposal area carry a higher heavy metal burden than those outside the dumping area, although often the differences are not very great.

The conclusions drawn to date from this study are that the sewage sludge is spread fairly widely after disposal and that although there has been a limited build-up of organic matter and heavy metals there is little evidence of gross pollution, either in chemical or biological terms.

The area which has been the subject of most intensive study is Liverpool Bay, where for over seventy years sewage sludge from the Manchester area has been dumped. The main reason for this intensive study, which began in 1969 and the initial stages of which were reported on in 1972(2, 17), was the need to predict what would happen if the existing rate of disposal (around ½ million wet tons/year) was increased by about 10 times. A multi-disciplinary working party was established and money was made available by the Department of the Environment, which acted as a coordinating body.

The study included investigations of the hydrography of the area, the composition of the sediments, movement of the sludge, the fisheries, the benthic fauna composition, etc. Toxicity tests were conducted using a variety of marine species, ranging from algae to shrimps and fish. Current measurements suggested a persistent two-layer structure of the water column, with a bottom onshore-moving residual current and a variable surface current, with a possible gyre action in the summer months.

Examination of the sediments in the survey area showed the highest accumulation of organic matter to be at the mouth of the Mersey, inshore from the dumping area. There was a strong correlation between the organic content and the contents of metals such as copper and zinc. The ratio was however different from that in the sewage sludge. There was little evidence of accumulation of sludge material in the immediate dumping zone.

The tracer experiments confirmed the tentative conclusions from these findings. The release of sewage sludge labelled with silver-110m was made late in September 1970. Subsequent surveys showed a rapid dispersion of the solids: by mid-October there was a distinct indication of movement towards the Mersey estuary and extensive dispersion. Further surveys confirmed these early trends. Up to one month after release no silver-110m could be detected on the shore but by early February 1971 most of the shore detection sites showed some silver-110m present, although only in small amounts. The final stage of this work was a projection of the distribution likely to take place when increased quantities are dumped. This showed that for one year's input the highest concentration to be expected was 0.5% in the sediments at the entrance of the estuary of the Mersey, but around the actual dumping site it was only 0.1%. The early surveys showed that the area of maximum fall-out would be a band approximately 8 x 30 km, running roughly east-west. On the basis of these surveys it was estimated that, although the sludge does contribute to the organic loading of the sediments, the existing rates of disposal contribute less than 10% of the total organic matter, the remainder coming from the estuary and from other terrestrial sources. Dispersal during neap tides was less pronounced and it was recommended that the dumping area be enlarged with vessels dumping along north-south lanes.

The report recommended that there would be little effect on the fauna, fisheries and sea bed composition if the quantity of sewage sludge were increased as proposed. A plan for further surveys was recommended and work is now in progress. The areas of disposal have been altered in accordance with the recommendations made in the light of the radio-tracer surveys.

Summary and conclusions

The disposal of sewage sludge by dumping at sea has been conducted from the United Kingdom for well over seventy years. The quantities of sludge disposed of in this way are steadily increasing, both in volume and with regard to the number of sources. Although in some areas limited accumulation of both organic matter and persistent substances has occurred, the extent to which this has taken place is small. The main reason for this limited effect is probably to be found in the adequate dispersion characteristics in relation to the quantities disposed of in the areas in question.

Disposal by dumping at sea of sewage sludges, and indeed wastes of industrial origin, from the United Kingdom will undoubtedly continue for the foreseeable future. However, the quantities which can be approved will continue to be related to the capacity of the area to receive wastes. The quantity can be estimated from a knowledge of current movements and the polluting characteristics of the waste, e.g. its oxygen demand, and its nutrient, metal and PCB content. Careful monitoring must of course be conducted to ensure that these predictions are accurate and that the capacity of the area to receive a waste is not pushed beyond its limit.

From this it should not be assumed that the sea may be used for the disposal of any waste which proves difficult or expensive to dispose of on land. Nor should the total capacity of an area to receive wastes be fully utilized. There must be some reserve of capacity to ensure that other uses of the area can continue. It is clear, however, that with care the sea can be used for the disposal of many waste materials and that in some respects it may offer advantages over land disposal, as currently conducted, although as new methods of treatment recovery or disposal (e.g. by incineration) are developed these may in turn be adopted in preference to sea disposal. At the present time, however, sewage sludge is a bulky waste with a high water content which proves difficult to dispose of on land but which with care can often be safely disposed of to sea.

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Table 1 Quantities of sewage sludge from England and Wales approved by MAFF for disposal on continental shelf since 1971

Area	Thousand tons per annum
Irish Sea	2 025 including Glasgow
Bristol Channel	356
English Channel	210
North Sea	5 463
Total	8 054

Table 2 Composition of UK sewage sludges dumped at sea

Component	Range	Mean
Solids - total (%)	1.3-60	8.4
Solids - volatile (%)	50.0-80	66
BOD (ppm wet weight)	40 000-290 000	230 000
Nitrogen*	0.1-48	12
Phosphorus*	0.3-20	6.0
Grease/fat*	0.1-26	8.4

*Given as percentage of dry solids.

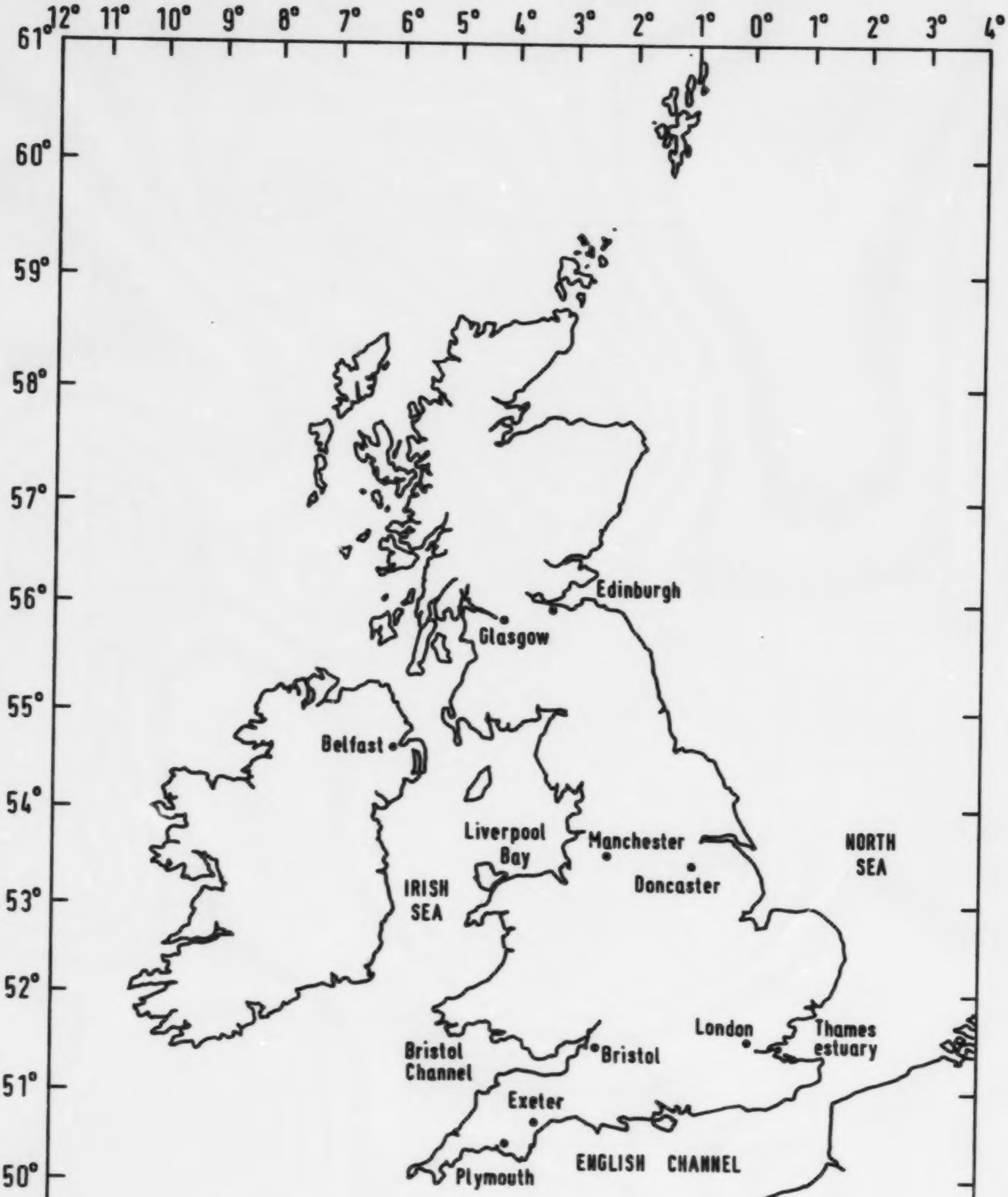
Table 3 Heavy metal and organochlorine content of UK sewage sludges dumped at sea

Component	Wet weight (mg/kg)		Dry weight (mg/kg)	
	Range	Mean	Range	Mean
Cadmium	0.2 - 6.3	1.1	3.0- 76	21
Chromium	0.3 - 152	21	2.0- 3 700	433
Cobalt	-	7.5	-	89
Copper	3.6 - 526	54	26.0-11 955	1 004
Lead	0.2 - 171	26	5.0- 1 208	361
Manganese	0.8 - 128	35	10.0- 1 530	423
Mercury	0.01- 20	0.35	0.2- 85	7.0
Nickel	0.3 - 92	9.3	4.0- 1 100	111
Zinc	3.0 -2 130	176	93.0-48 410	3 409
Total organochlorine pesticides	0.01- 0.50	0.20	0.3- 9.4	3.0
Total PCBs	0.1 - 8.6	1.2	0.2- 94	20

Figure captions

Figure 1 The British Isles, showing the main areas referred to in the text.

Figure 2 North Sea, co-range chart.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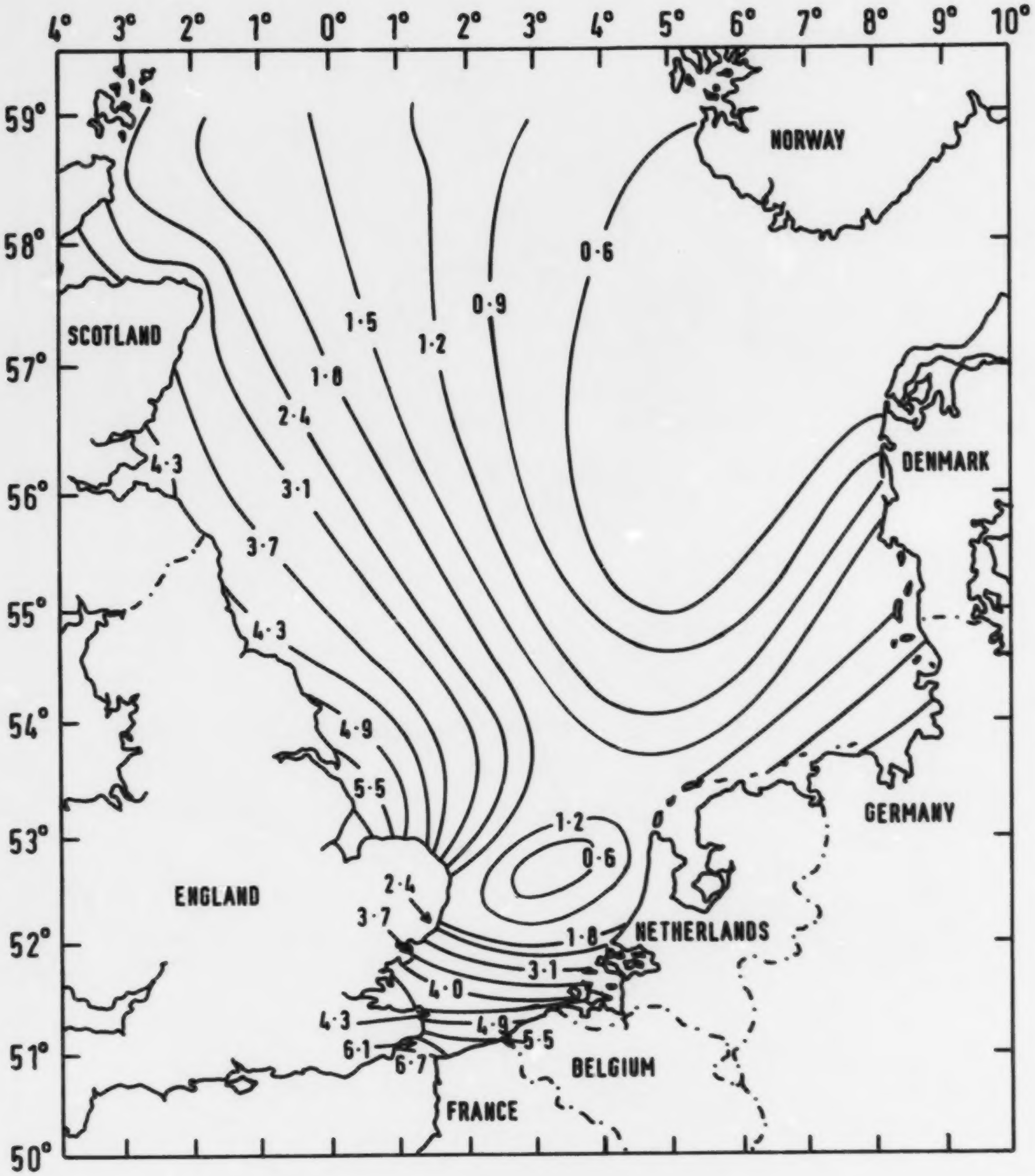
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Preliminary Summary
of
Sludge Degradation Studies
in a
Marine Benthic Environment

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Research Symposium
Pretreatment and Ultimate
Disposal of Wastewater Solids

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Rutgers University
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This paper describes in summary form, the research efforts of the author in the past two years. Details of each topic, a description of experiments in progress and analyses of all data will be submitted in dissertation form as partial fulfillment of the requirements for the degree of Doctor of Philosophy in the Department of Civil Engineering, Oregon State University.

ABSTRACT

A series of field and laboratory experiments were carried out to determine degradation rates of organic sludge deposits in a sea floor environment. In laboratory experiments total carbon concentrations were found to decrease approximately 70% in 80 days for experiments at 1 atmosphere. The total carbon decreased 30% in 47 days, in elevated pressure experiments with a Deep Sea Simulator. Oxygen consumption rates ranged from 3.74 grams-O_2/m^2/day to 0.55g-O_2/m^2/day depending on sludge source, type of digestion and time since deposit. Dissolved oxygen was depleted beyond 1 millimeter in the bed; oxidation-reduction potential ranged from +150 millivolts at the sludge bed surface to -100 mv at 2 centimeters. Diffusion of hydrogen sulfide into underlying sediments began at 24 days into the experiments and proceeded at a rate of 0.09 mm/day. Due to compaction, sludge beds decreased to one-half their original thickness in 80 days. Approximately 60% of all compaction occurred in the first day.

Current experiments are being devoted to better definition of initial carbon losses and documentation of carbon decreases in a 50 centimeter bed. A conceptual mathematical model of dissolved particulate and gaseous fractions of sludge bed carbon is being formulated in an effort to better understand the mechanisms and transfer rates of the sludge bed carbonaceous materials.

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INTRODUCTION

Significant amounts of domestic sewage sludges are being released to coastal waters without an adequate understanding of the impact of such practices on marine ecosystems (Buelow, 1968; Brown, et. al., 1971; National Marine Fisheries, 1972). Over 1.3 million metric tons of sewage sludge are produced annually in coastal regions of the United States. Due to population increases in these areas, this figure is expected to exceed 1.9 million metric tons annually by the year 2000 (Council on Environmental Quality, 1970).

Additional disposal requirements will be created by upgrading of existing domestic treatment facilities to secondary, and primary treatment of raw sewage discharges. In 1970, more than 3.8 million cubic metres of sludge were dumped 22 kilometres from the New York Harbor entrance. By last year, the amount had risen to 4.4 million cubic metres (Dewling, 1974). With further wastewater treatment requirements, the annual disposal requirements of the New York-New Jersey metropolitan area are expected to exceed 11.5 million cubic metres (Horne, et. al., 1971; Gross, 1969).

Because of the probable future increases in the ocean disposal of sludges, the need to optimize disposal techniques has become even more critical. Such an optimization program cannot be developed without a sound scientific basis. Yet data on the fate and effects of disposed sludges is lacking. A Smithsonian Institution evaluation of New York Bight waste disposal studies concluded that research on post-deposition waste characteristics is preliminary and inconclusive (Buzas, et al., 1972). The report stressed the need to determine the time required to stabilize and decrease the extent of benthic deposits on the ocean floor. Previous descriptions of equilibrium conditions in ocean disposal zones, have assumed decomposition-rate constants from river deposit data (Barnett, et al, 1969).

The primary objective of this research was to determine the rate and extent of organic carbon decreases in a digested domestic sewage sludge deposited on the sea floor. A set of predictive mathematical equations was formulated to describe decreases in the organic content of a sludge deposit as a function of time and depth in the bed. A sludge-bed model is being developed to describe temporal and spatial exchange rates between dissolved, particulate and gaseous phases of the sludge bed. Experimental observations have provided additional data on compaction rates, losses of volatile fractions during settling through a sea water column, sulfide diffusion rate into underlying sediments and sludge oxygen consumption rates.

BACKGROUND

Previous laboratory studies on the stabilization rates of organically laden river deposits were devoted to the measurement of the oxygen depletion of water in contact with deposits of specific configuration for a known period of time (Fair, et. al., 1941). The oxygen consumption was partly attributed to oxidation of organic matter close to the sludge bed surface. Exposure of reduced anaerobic degradation products, forced upward during compaction, was also thought to impose a substantial oxygen demand on bottom waters. Allowance was given for the anaerobic production of gases (methane and carbon dioxide) which could, upon accumulation, become sufficently buoyant to erupt through the deposit causing significant overturn, exposure of reduced sediments and resultant high oxygen utilization rates in near bottom waters.

Comparison of benthal oxidation rates as measured above with the anaerobic stabilization rates obtained by running a time series of 5-day BOD tests on samples from a deposit covered with oxygen depleted water showed the anaerobic rates to be much higher (Rudolfs, 1938; Mohlman, 1938). The conclusion was that the rate of oxygen consumption by a deposit does not necessarily correlate with the overall stabilization rate due to different chemical conditions at the surface and deeper in the deposit. Reduced by-products in the deeper anaerobic regions may not exert a predictable oxygen demand on the overlying waters.

Kuznetsov (1973) suggested that the amounts of released carbon dioxide and methane would better relate overall sediment decomposition rates since these gases are created during mineralization in anaerobic and aerobic regions. Simultaneous measurement of oxygen consumption in the overlying waters, calculation of the equivalent CO_2 produced due to the aerobic degradation, and subtraction of this from the total CO_2 produced would provide measure of the CO_2 associated with anaerobic degradation.

Previous research on organic deposit stabilization has been primarily devoted to measuring the effects of such a process on overlying waters. Such an approach is particularly attractive because low oxygen levels in these waters can significantly affect both pelagic and benthic biota. The studies revealed the differences in aerobic and anaerobic breakdown rates, but an effective procedure to relate this information to the overall (aerobic and anaerobic) stabilization rate of a river bed or sea floor deposit has not been developed.

The presence or absence of molecular oxygen as a terminal hydrogen-ion acceptor is one of many parameters which could control the microbial conversion of organic matter. Substrate availability, physical stability of the deposit, varying chemical conditions and bacterial community succession could each be influential.

The size of the particulate fraction of sewage has also been shown to influence the biochemical oxidation rate (Balmat, 1957). Laboratory studies over a period of 5 days showed that aerobic decomposition of relatively large settleable and supracolloidal (1-100 micron) solids may be limited by a slow hydrolysis rate, whereas colloidal (0.08 to 1.0 micron) and dissolved (<0.08 micron) organic matter is utilized much more rapidly. Biochemical oxygen demand (BOD) rate constants (K_1) defined by the following equation were found to range from 0.18 to 0.90 (per day).

$$\frac{L_t}{L} = e^{-K_1 t} \qquad (1)$$

where L = ultimate biochemical oxygen demand (ppm)
L_t = the BOD at time t (ppm)
K_1 = rate constant (per day)

The rate studies on sewage solids suspended in mineral dilution water (with a 1% sewage seed) were made at 20°C over a 5 day period with a Warburg Respirometer.

LABORATORY MEASUREMENTS

Microbial degradation of settled sludge on the sea floor involves solubilization and subsequent utilization of dissolved organic and inorganic compounds for the production of energy. In aerobic zones, organic or reduced inorganic compounds are oxidized to carbon dioxide and used for cell synthesis, maintenance, motility or active transport. At sludge bed depths greater than about 1 millimeter, molecular oxygen is rapidly depleted and degradation proceeds either anaerobically by anaerobic respiration in the presence of exogenous hydrogen-ion acceptors (e.g., sulfates and nitrates), or by fermentation where lack of exogenous acceptors requires their endogenous generation (Schroeder et al., 1966). As a result of these metabolic processes, the sludge organic carbon content will continually decrease due to conversion to inorganic products or gaseous losses. The rate of breakdown, however, will be dependent on the dominant degradation mode at different locations in the bed. Since carbon is the most abundant element in sewage sludge (Gross, 1970) and carbonaceous compounds act as food sources for resident bacteria, large changes in sludge carbon concentrations will occur as decomposition proceeds. For this reason, organic and inorganic carbon were selected as principal indicators of the stabilization process.

Initial laboratory experiments have been devoted to the measurement of sludge sample total carbon (TC) and total inorganic carbon (TIC) concentrations as a function of time since deposition and depth in the bed. Organic carbon content was calculated as the difference between these two measurements. An OIC* Direct Injection Module was used for TC determination and TIC measurements were made by acidifying samples in the presence of nitrogen gas and recording the amount of CO_2 released. Sludge carbon content was determined on samples from treatment plant holding tanks just prior to loading on an ocean going barge, after settling through sea water, subsequent to transfer of the settled solids to a sea floor simulator and at selected time intervals thereafter (30 and 60 days on initial experiments; in a geometrical time sequence on those currently in progress). Aliquots of all samples were also analyzed for total solids, total volatile solids and ash content.

Parameters measured to record the sludge chemical characteristics included sulfate, phosphate, nitrate, and Kjeldahl nitrogen. Profiles of oxygen concentration, oxidation-reduction potential and pH in the experimental apparatus were also made. Microbial observations were made and photographed at 1400X using phase contrast microscopy.

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*Oceanography International Corporation, Mod. 0524B-HR. Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

SIMULATION PROCEDURES

Initial laboratory experiments were designed to measure approximate rates of organic carbon conversion and changes in chemical conditions in the bed. Oxygen utilization rates were also measured to compare with respirometric recordings made during field studies.

Experiments of 42 and 81 days were carried out with the one-atmosphere acrylic reactor shown in Figure 1. Sea water was pumped from a 95 liter (25 gallon) reservoir, through a series of baffles designed to insure uniform channel flow over the bed, and returned to the reservoir. Reservoir seawater was continuously replenished (5 day residence time) from a 3800 litre storage tank. Dissolved oxygen concentration was maintained at 6-7 mg/l and flow rate controlled to provide a mean current of 1.2 cm/sec. Fifty millilitre sludge samples were periodically withdrawn with a 13 gauge needle through a series of sampling septums along the reactor sidewall.

Two 47 day experiments at 6.8 atmospheres (100 PSI) and 34 atmospheres (500 PSI) were conducted with the Deep Sea Simulator shown in Figures 2 and 3. A sludge reactor similar to that described above was sealed inside the pressure vessel.* Sea water was pumped by an air-hydraulic pump through the end-cap into the reactor, over the sludge bed and out into the pressure vessel interior. An exit port in the opposite end-cap permitted sea water to then return to the reservoir. The pressure pulse caused by the piston-pump was reduced through the use of an accumulator at the inlet port. Flow rate and internal pressure were controlled by regulating line air pressure (pump rate) and exit valve settings. Although initial experiments were at pressures of less than 35 atmospheres (to simulate continental shelf and inner continental slope site pressures), the simulator system is designed to operate up to 680 atmospheres (10,000 PSI).

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*Converted surplus U.S. Navy, MK13, Mod. 2, High Capacity 406m (16-inch) projectile.

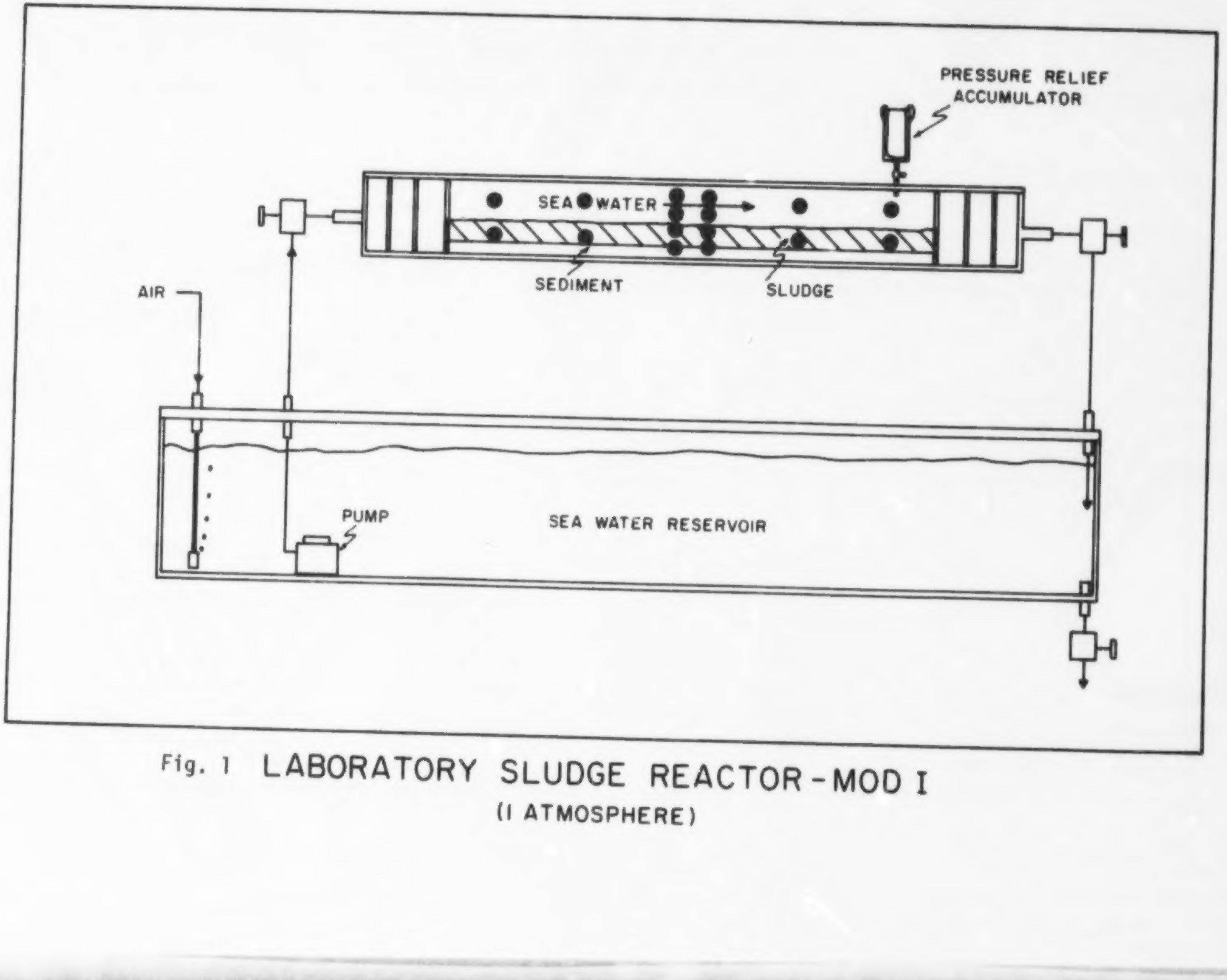
PRESSURE RELIEF
ACCUMULATOR
SEA WATER
SEDIMENT
SLUDGE
AIR
PUMP
SEA WATER RESERVOIR

Fig. 1 LABORATORY SLUDGE REACTOR - MOD I
(1 ATMOSPHERE)

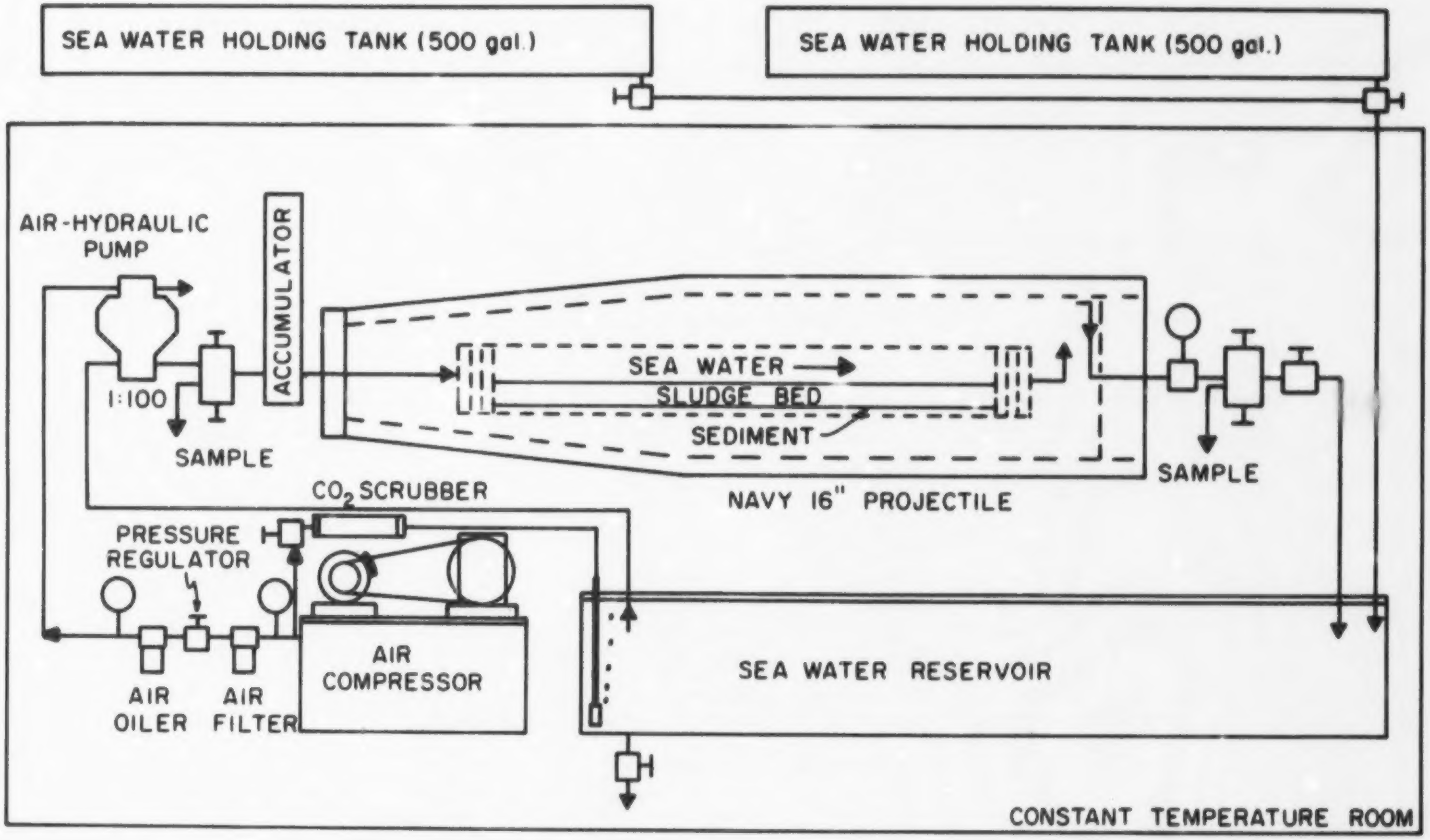
SEA WATER HOLDING TANK (500 gal.)
SEA WATER HOLDING TANK (500 gal.)
AIR-HYDRAULIC PUMP
1:100
SAMPLE
ACCUMULATOR
SEA WATER
SLUDGE BED
SEDIMENT
NAVY 16" PROJECTILE
SAMPLE
CO_2 SCRUBBER
PRESSURE REGULATOR
AIR OILER
AIR FILTER
AIR COMPRESSOR
SEA WATER RESERVOIR
CONSTANT TEMPERATURE ROOM

Fig. 2 DEEP SEA SIMULATOR

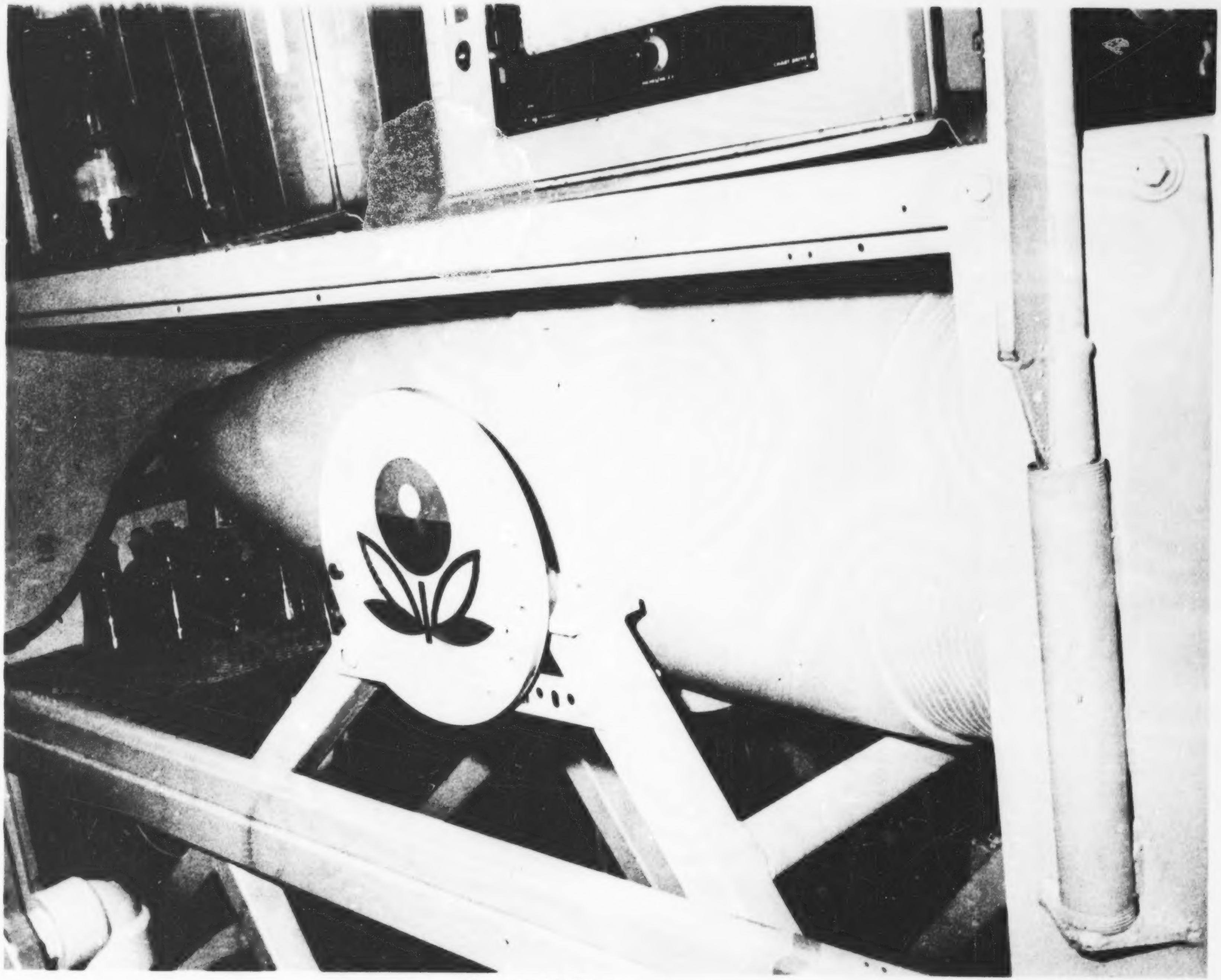

Fig. 3 Simulator System

Because of reports that controlled decompression and recompression rates can minimize adverse effects on bacteria (Zobell, 1970) non-selective sludge sampling through the pressure vessel end-cap was discarded in favor of opening the vessel and removing the reactor. After samples were selectively taken through the side-wall septums the reactor was returned to the pressure vessel and slowly recompressed to the experimental pressure.

Dissolved oxygen concentrations in sea water overlying shallow (2 cm.) sludge beds were measured using a sealed glass reactor equipped with an IBC* dissolved oxygen sensor with stirrer (Figure 4). Aerated seawater was continuously fed into and removed from the reactor to provide required nutrients and to maintain a 6-7 mg/l oxygen concentration. Air and sea water supplies were periodically stopped and the changes in the oxygen content of the entrapped water was recorded on a strip chart recorder. The dissolved oxygen probe was calibrated prior to each set of measurements by using the Azide Modification of the Winkler Titration described in Standard Methods (APHA, 1971).

LABORATORY CARBON DATA

Tables 1 through 5 in Appendix list the concentrations of total carbon, total inorganic carbon, total solids, total volatile solids and ash content of samples from the three experimental systems previously described. Total organic carbon is calculated. Due to initial compaction, the solids (and hence carbon) concentrations at a given location in the bed will increase. To visualize only those changes in carbon concentration due to other causes, the data are presented in normalized form as mg-C/gram solid.

In the first experiment, the sludge was not pre-settled in seawater prior to placement in the sludge-bed reactor, hence a relatively high TC

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*International Biophysics Corporation, Mod. 501-001

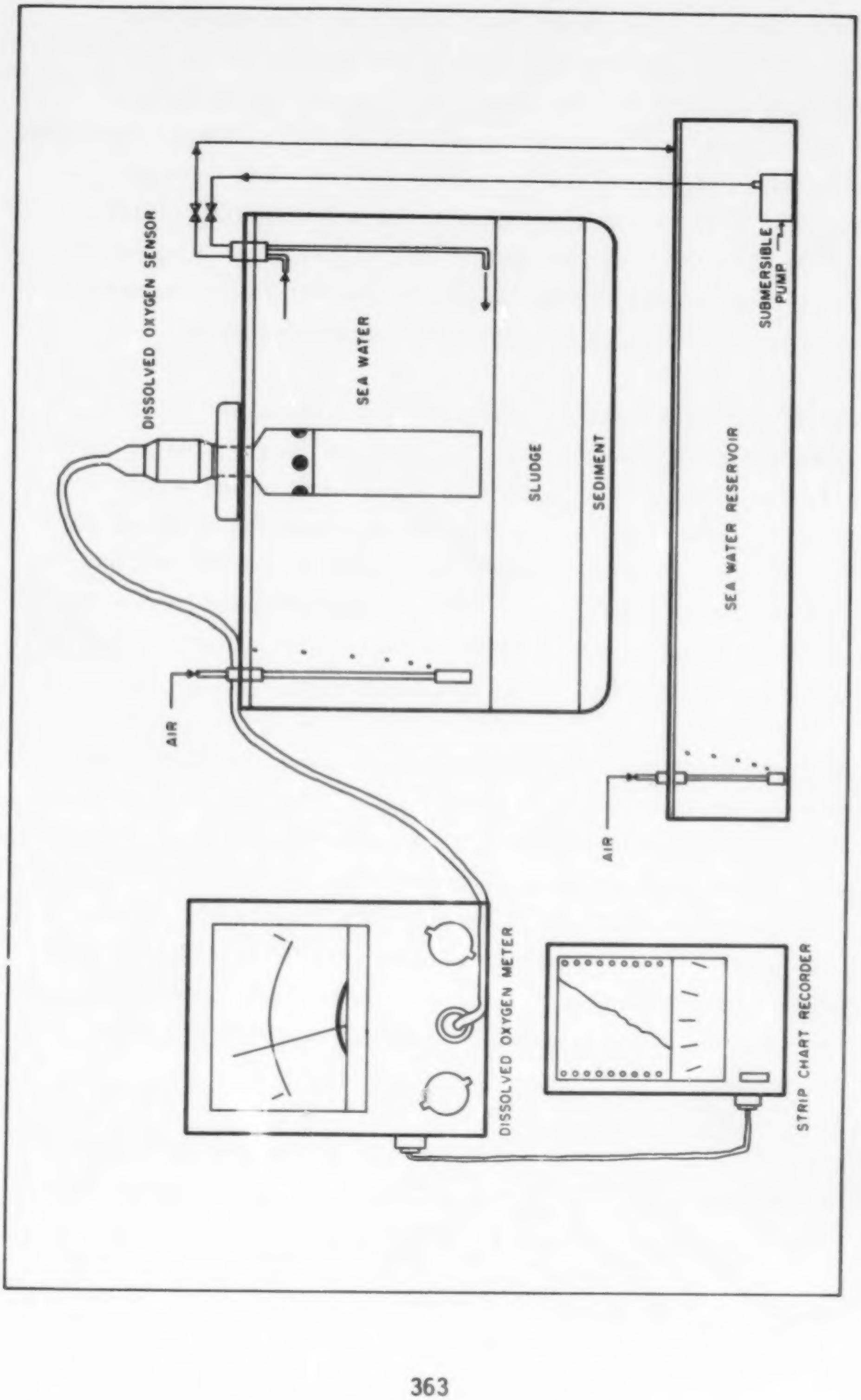
DISSOLVED OXYGEN SENSOR
SEA WATER
SLUDGE
SEDIMENT
AIR
SEA WATER RESERVOIR
SUBMERSIBLE PUMP
AIR
DISSOLVED OXYGEN METER
STRIP CHART RECORDER

Fig. 4 LABORATORY SLUDGE RESPIROMETER

of 329.5 mg-C/g solid. Presettling through a 0.75 metre water column was carried out for all other experiments resulting in a relatively large decrease in total carbon content to approximately 120 mg-C/g solid. Careful assessment of such losses is currently being made by analysis of both sludge and seawater before and after settling. Parameters determined include total and dissolved carbon (organic and inorganic), total and volatile solids, and ash content. An understanding of the form and exchange of sludge organic matter and carbon after dumping in sea water is critical to the prediction of concentrations in the waste upon arrival on the sea floor and subsequent degradation rates.

The data in Tables 1 through 5 indicate that, where sea water continually moved over the sludge bed, TC concentrations decreased more rapidly and to a greater extent in surface zones than in the deeper anaerobic regions. Some reservation is made regarding the value of the surface carbon in that it may represent the characteristic carbon concentration in the bacterial slime layer. This is substantiated by the fact that surface carbon concentrations remained higher for surface sludge in the glass (static) reactor where no slime layers developed.

Figures 5 and 6 (plotted data from Tables 1 and 5) indicate that decreases in total carbon content of subsurface sludge amounted to 71 and 68%, respectively, for 81 and 68 day experiments. Studies of organic river deposits showed that in 45 to 60 days 50% of the ultimate oxygen demand (L) was expended. In 7 to 10 months, this requirement reached 90% of L (Fair et al, 1941). Because measured initial degradation rates were high, experiments in progress involve a geometrical sampling pattern (0, 1/4, 1/2, 1, 2, 4, 8, 16 days etc.) to better document the first stages of degradation.

A comparison of the differences in initial carbon concentrations (Figures 5 and 6) and associated slopes of the Total Carbon versus Time

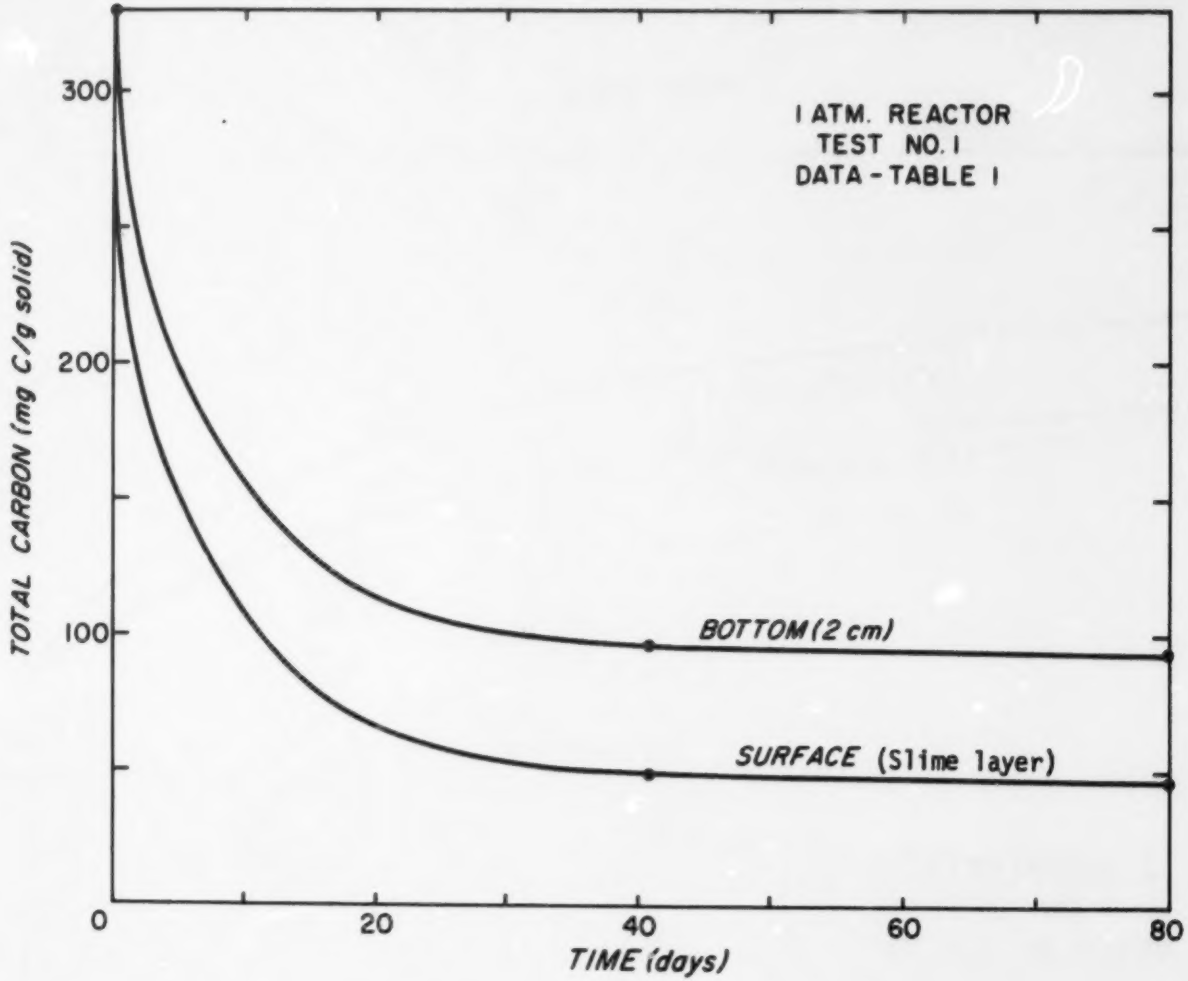

Fig. 5 Total Carbon vs. Time
(1 atmosphere reactor)

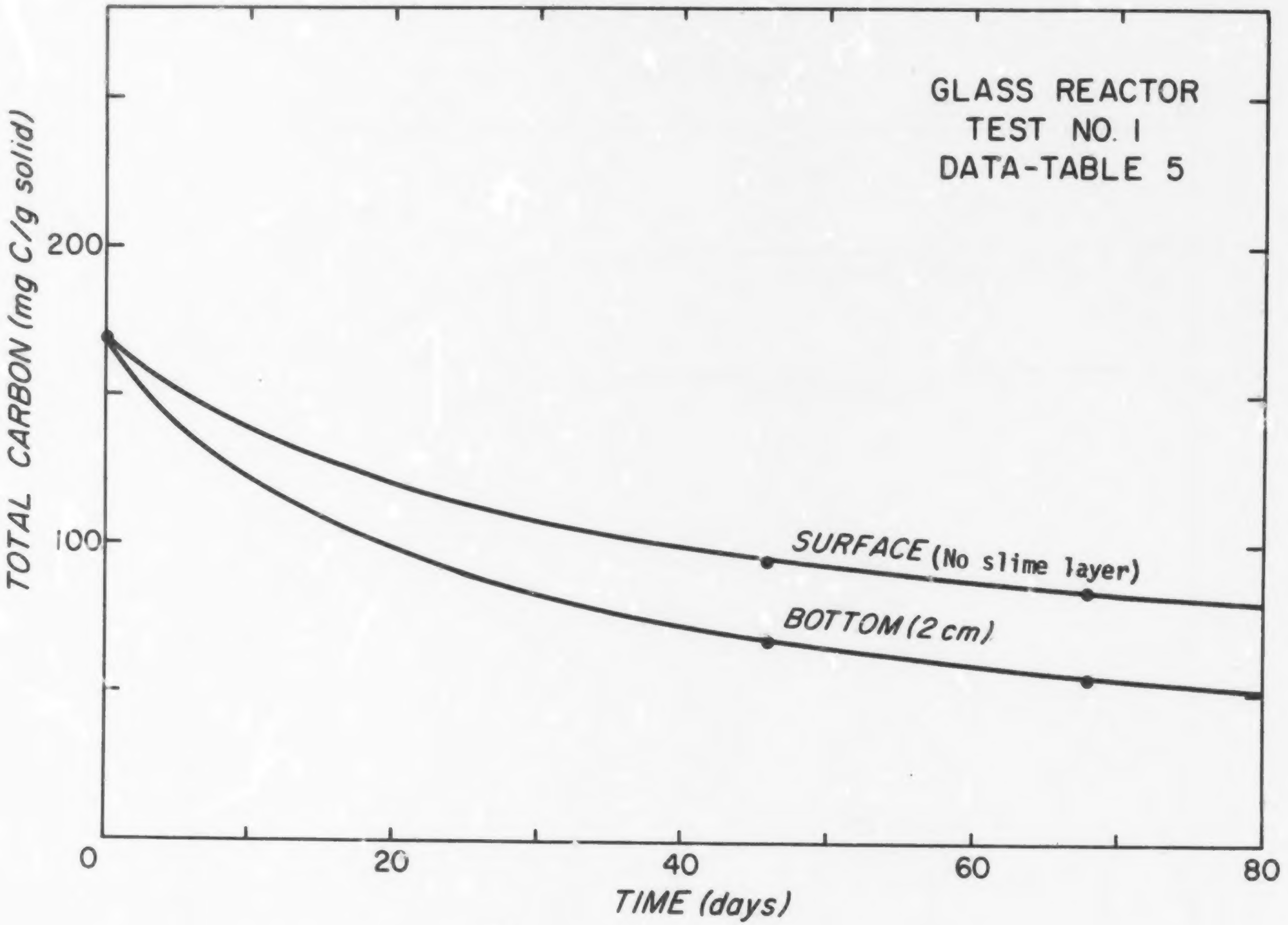

Fig. 6 Total Carbon vs. Time
(laboratory respirometer)

plots at these points suggests that the rate of change of carbon mass is proportional to the mass present. This might be described by a first order equation, viz.:

$$\frac{dC}{dt} = -KC \qquad (2)$$

where

C = Total carbon mass per unit volume in deposit at any time t

K = Reaction velocity constant

Using the solution to equation (2) in the form of

$$K = \frac{1}{t} \ln \frac{C}{C_o} \qquad (3)$$

where

C_o = Initial total carbon mass per unit volume

the reaction velocity constants calculated for each experiment are given in Table 6. These values can be compared to sludge benthal oxidation velocity constants measured by Fair, et al. of 0.007 to 0.010 (per day) and sludge stabilization rate constants by Rudolfs and Mohlman of 0.010 and 0.012 (per day). Work by McGowan, Frye and Kershaw on English river muds produced reaction constants of 0.001 to 0.005 (per day) (McGowan, et al., 1913).

Although much more data is required, some insight into the degradation rates and variation in these rates can be gained from data obtained to date. There appears to be a decrease in the reaction velocity under

Table 6 REACTION VELOCITY PARAMETERS (K)

$\frac{dC}{dt} = -KC$	Flowing 1 Atmosphere Reactor		Flowing Pressure Reactor		Semi - Static Glass Reactor	
	Surface	2 cm.	Surface	2 cm.	Surface	2 cm.
EXPERIMENT I	0.046 (41) 0.024 (81)	0.030 (41) 0.015 (81)	100 PSI 0.017 (47)	0.006 (47)	0.013 (46) 0.010 (68)	0.020 (46) 0.017 (68)
EXPERIMENT II	0.015 (42)	0.011 (42)	500 PSI 0.016 (47)	0.007 (47)	———	———

* Nos. in parentheses designate time (days) into the test.

pressure conditions (K = 0.006, 0.007). Jannasch (1971) has shown that microbial degradation rates of selected organic substrates in the deep ocean (5000 metres) are 10 to 100 times slower than at 1 atmosphere and similar temperatures. Paul, et. al. (1971) attributes the decreased microbial metabolic rates to pressure (and low temperature) inhibition of cellular acid uptake.

In tests where the reaction velocity is evaluated at two points in time, K appears to have decreased implying that a modification to equation 2 must be made to appropriately describe the carbon changes. Fair points out that the constant K describes the initial availability of the substrate but does not describe the reduction in this availability with time. He suggests the inclusion of a retardation factor of the form

$$\frac{dC}{dt} = -\left(\frac{K}{1 + at}\right)C \qquad (4)$$

where

a = Coefficient of retardation which is dependent on type of organic matter present and condition of the environment

Whether this equation or one of another form describes carbon changes in an ocean sludge bed will become apparent by analyses of data from detailed experiments now in progress.

LABORATORY OXYGEN CONSUMPTION RATES

Initial oxygen uptake of a 2.0 cm bed of anaerobically digested sludge* (measured with the apparatus in Figure 4) was 2.72 g-O_2/m^2/day. After 53 days exposure to aerated sea water, this rate had decreased to 0.95 g-O_2/m^2/day. Anaerobically digested sludge from a New York sewage treatment plant** required 0.72 g-O_2/m^2/day initially, decreasing to 0.55 g-O_2/m^2/day in 20 days.

OTHER LABORATORY MEASUREMENTS

Prior to concentrating on measurements of carbon changes in a sludge bed, some data on organic nitrogen and phosphorus content of the bed were also collected (Table 7). In all experiments, there was a considerable decrease in the total dissolved and suspended organic nitrate and phosphate content suggesting biochemical conversion and/or transfer to the sea water (i.e., solubilization and subsequent loss due to compaction).

Redox-potential (E_h) profiles on the shallow beds showed a typical change from +150 millivolts at the bed surface to -100 millivolts at 2 cm. deep. Current experiments with a redesigned reactor will permit E_h profiles to bed depths of 50 cm.

The diffusion of hydrogen sulfide into the sediments beneath the sludge could easily be seen due to the downward progression of a dark black zone in which iron sulfide was formed. The discoloration was typically detected 20 to 24 days after start of the experiment and progressed downward at a rate of 0.09 mm/day. At the conclusion of an 84 day experiment the Fe-S band was 7 mm in thickness.

* Corvallis Wastewater Treatment Plant, Corvallis, Oregon

**Nassau County Sewage Treatment Plant, East Rockaway, New York

TABLE 7 Laboratory Nutrient Measurements

SLUDGE NITROGEN and PHOSPHORUS (g./kg.dry solid)		One atmosphere reactor (NYK)					Pressure Reactor (NYK)			Semi-static Reactor (CORV)				
		As Received	T_o + 41		T_o + 81		As Received	T_o + 47		As Received	T_o + 46		T_o + 68	
			Surf	2 cm.	Surf	2 cm.		Surf	2 cm.		Surf	2 cm.	Surf	2 cm.
NITROGEN	Total Kjeldahl	52.10	10.40	13.30	7.84	11.50	42.00	10.70	17.10	26.50	13.00	9.32	13.00	7.05
	NH_3 - N	12.60	0.63	0.70	0.40	0.58	3.40	0.33	0.44	3.71	0.49	0.38	0.46	0.38
	Organic - N	39.50	9.77	12.60	7.44	10.92	38.60	10.37	16.66	22.79	12.51	8.94	12.54	6.67
PHOSPHORUS	Total	16.40	3.11	3.54	2.61	2.98	19.40	4.50	6.75	11.55	4.89	3.79	4.73	2.67
	Hydrolyzable + Ortho	13.30	2.28	2.09	2.07	—	16.10	3.20	4.64	9.03	3.73	2.68	3.42	2.42
	Organic - P	3.10	0.83	1.45	0.54	—	3.30	1.30	2.11	2.52	1.16	1.11	1.31	0.25

Due to compaction of the settled solids, experimental sludge beds decreased to approximately 1/2 their original thickness during the 60 to 80 day tests. Initial rates are very high (i.e., 54 to 56% compaction* within the first 12 hours). At the conclusion of one day, compaction is typically 60% complete. Thereafter, decreases in bed thickness occur very slowly at a rate of 1.2mm/day.

Pratt _et. al._ (1973) gave a description of microbial colonies formed on the surface layers of sludge during laboratory experiments. The experiments described in this paper confirmed their observations that an initial predominance by vibrio-like rods was followed by a succession of trichome-producing bacteria possibly of the genera _Thiothrix_ and _Beggiatoa_ (Breed, _et. al._, 1957). In the pressure experiments, filamentous bacteria also predominated the surface slime layer, but the encased granules did not occur in long chains (when observed after a 2 hour decompression from 34 atmospheres). Short segments contained typically 3 or 4 granules. Zobell _et. al._, (1950) found that some barotrophic organisms modify their morphology to accommodate higher pressures and that changes can be expected in some cases upon decompression. No dominance of a single species of bacteria occurred deeper in the sludges at either one atmosphere or under pressure.

FIELD STUDY

A four week field study was devoted to a preliminary assessment of sludge dispersion and degradation characteristics upon release to offshore waters. The research site, located in 15 metres of water, was instrumented with current meters, water quality sensors and station markers (Figure 7). A baseline survey included areal sampling of sea water and bottom sediments to establish pre-release carbon, nitrogen, phosphorus, and volatile matter levels. Anaerobically digested sludge released near the bottom at the upstream end of the marked site formed a patchy (0-1 cm. thick) sludge bed 12 metres wide and 60 metres in length.

*Compaction is defined as initial minus bed thickness at any time t, divided by initial minus terminal bed thickness at the experiment conclusion.

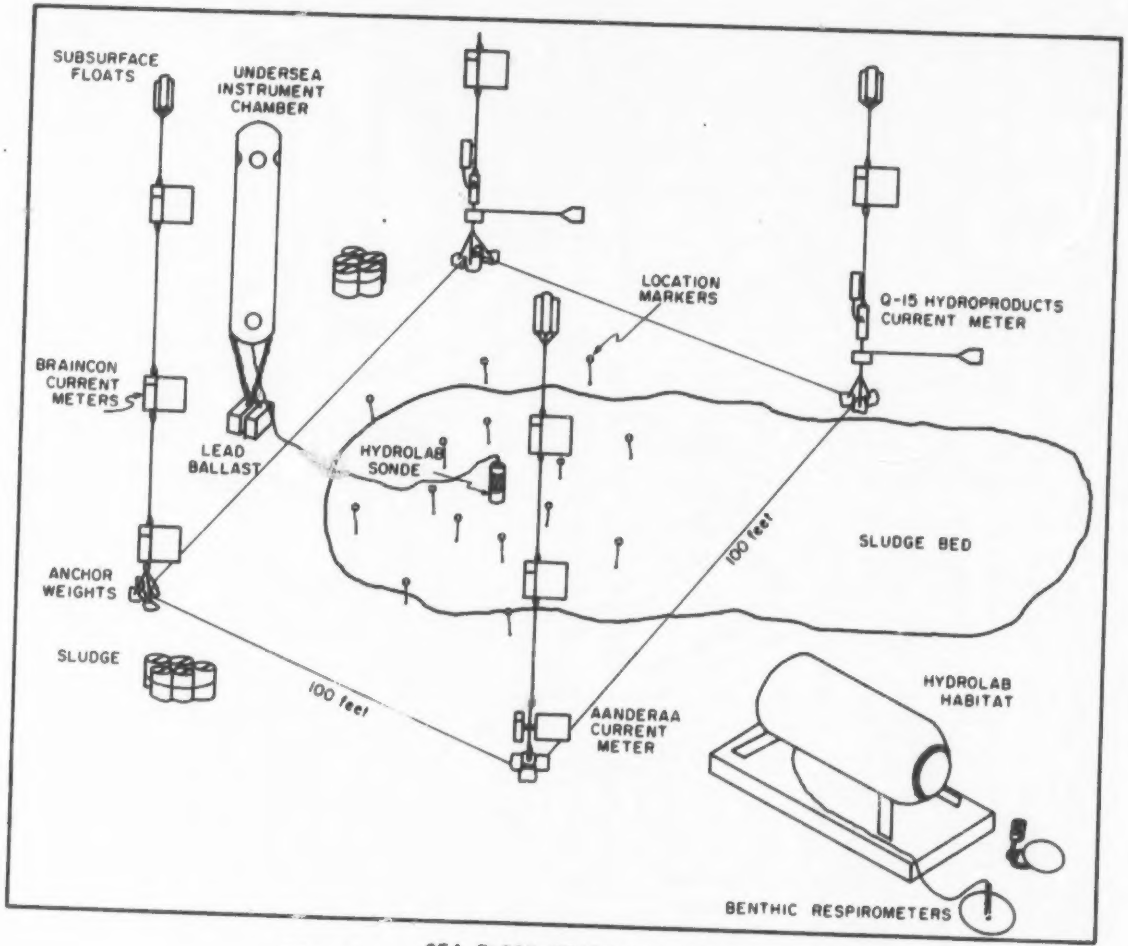
SUBSURFACE FLOATS
UNDERSEA INSTRUMENT CHAMBER
BRAINCON CURRENT METERS
LEAD BALLAST
ANCHOR WEIGHTS
SLUDGE
HYDROLAB SONDE
LOCATION MARKERS
Q-15 HYDROPRODUCTS CURRENT METER
SLUDGE BED
100 feet
100 feet
AANDERAA CURRENT METER
HYDROLAB HABITAT
BENTHIC RESPIROMETERS

Fig. 7 SEA FLOOR RESEARCH SITE
Freeport, Grand Bahamas

Samples of the bed and near bottom waters were transported ashore for analysis and preparation for shipment. Heavy seas (estimated significant wave height of 3.5 metres) and bottom currents (to 0.5 knots) effectively removed the settled sludge within several days of the release.

All samples taken during the field exercise have not been analyzed and current meter data has yet to be completely reduced. Pre-release organic carbon concentrations in near bottom and interstitial waters ranged from 0.9 to 1.1 mg-C/liter. Samples taken in the plume contained 2.0 to 2.7 mg-C/l and post release bottom samples (in the bed zone) contained 3 to 20 mg-C/l. Within 6 days bottom and interstitial water organic carbon concentrations had returned to approximately 1 mg-C/l.

IN-SITU OXYGEN UPTAKE

Oxygen consumption rates of anaerobically and aerobically digested sludges were measured using bottom mounted 76 and 50 cm. diameter transparent acrylic hemispheres (Figure 8). The larger dome contained a temperature/dissolved oxygen probe with magnetic stirrer to insure adequate sea water flow over the sensor membrane. A 30 metre cable linked the sensor to a readout meter and strip chart recorder within the nearby habitat used as living quarters. As required, a submerged pump replenished the respirometer with oxygen laden seawater. The second hemisphere used a recirculation pump system to insure movement of seawater in contact with the sludge bed past an externally mounted oxygen probe chamber and back into the dome. The probe meter and strip chart recorder were mounted on the sea floor adjacent to the dome.

The oxygen utilization rate of aerobically digested sludge varied from an initial rate of 3.74 grams of O_2/m^2/day to 2.0 g-O_2/m^2/day after 65 hours. Anaerobically digested sludge oxygen use was initially 2.51 g-O_2/m^2/day and decreased to 1.68 g-O_2/m^2/day in 70 hours. Oxygen

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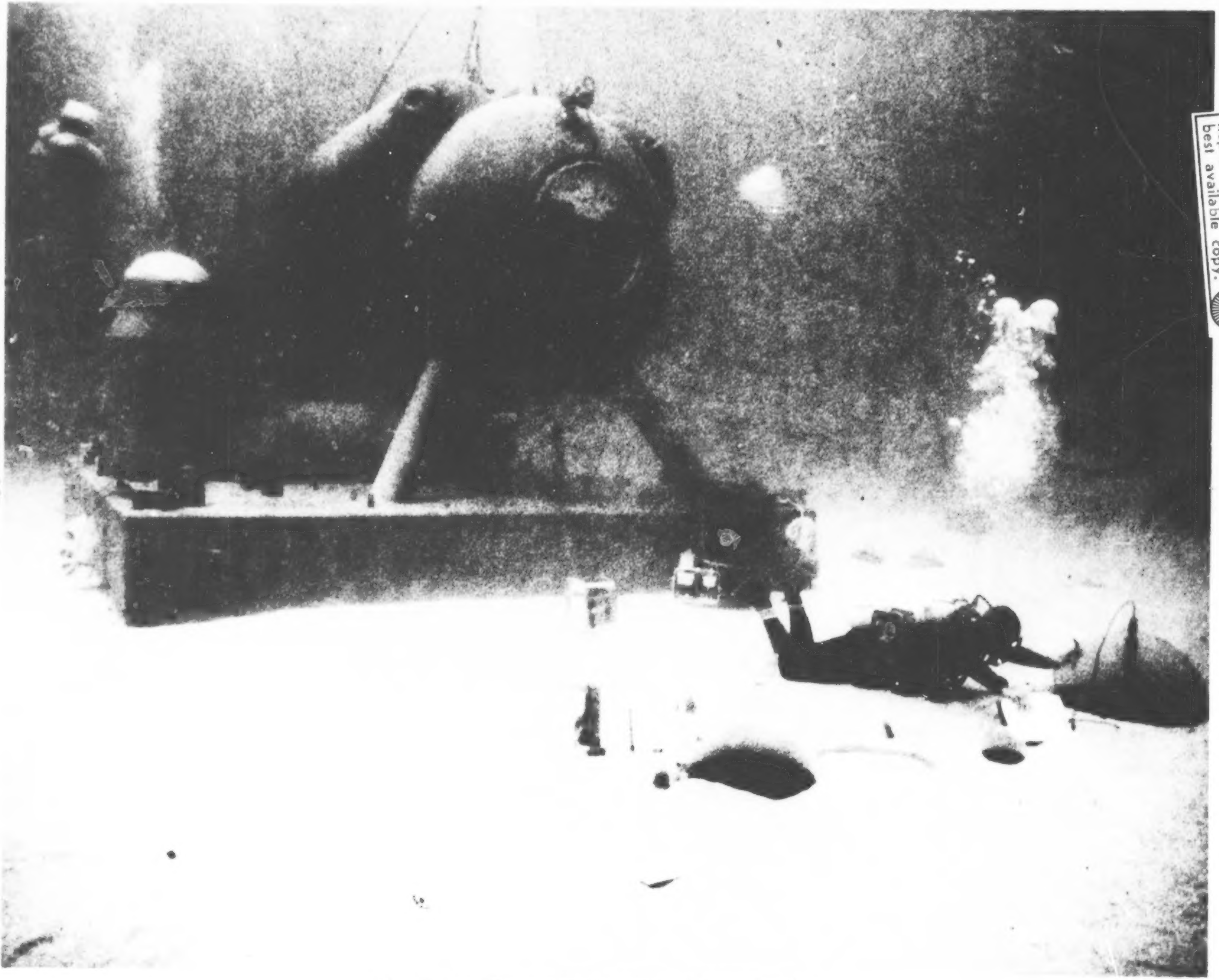

Fig. 8 Sea Floor Respirometer Measurements

uptake rates for marine sediments in Long Island Sound have been measured at 0.03 $g-O_2/m^2/day$ and for sediments near Woods Hole Oceanographic Institution rates ranged from 0.01 to 0.007 $g-O_2/m^2/day$ (Carey, 1967).

EXPERIMENTS IN PROGRESS

To account for the relatively large losses of carbon in the sludge bed, the amounts of carbon in the seawater moving over the bed and in gas emanating from the bed are also being monitored. By centrifuging (16,300 G's for 1 hour) and filtering (0.45 micron), a distinction is being made between total, dissolved and particulate carbon concentrations in the sludge. Such measurements give the added definition required to quantify solubilization rates, expulsion losses due to bed compaction, and conversion to dissolved and gaseous end products. A redesigned reactor allows more detailed sampling of sludge beds to 50 centimetres (Figure 9). A gas trap has been included and provisions for gas chromatographic analyses (carbon dioxide and methane) have been made. The sea water flowing over the bed is sealed from the atmosphere, saturated with carbon-dioxide free air and monitored for changes in carbon concentrations.

Although the forms of carbon in a domestic sludge prior to disposal can be fairly well specified, the changes in these forms during settling through sea water and during benthic degradation have yet to be determined in detail. Experiments in progress have been designed to partly fill this information gap. In addition, a preliminary (conceptual) sludge bed carbon model is being developed in an effort to better describe degradation and exchange mechanisms. The model allows for advective and diffusive transfer of particulate, dissolved and gaseous fractions of the sludge carbon between bottom waters, aerobic and anaerobic regions in the bed, and sediments beneath.

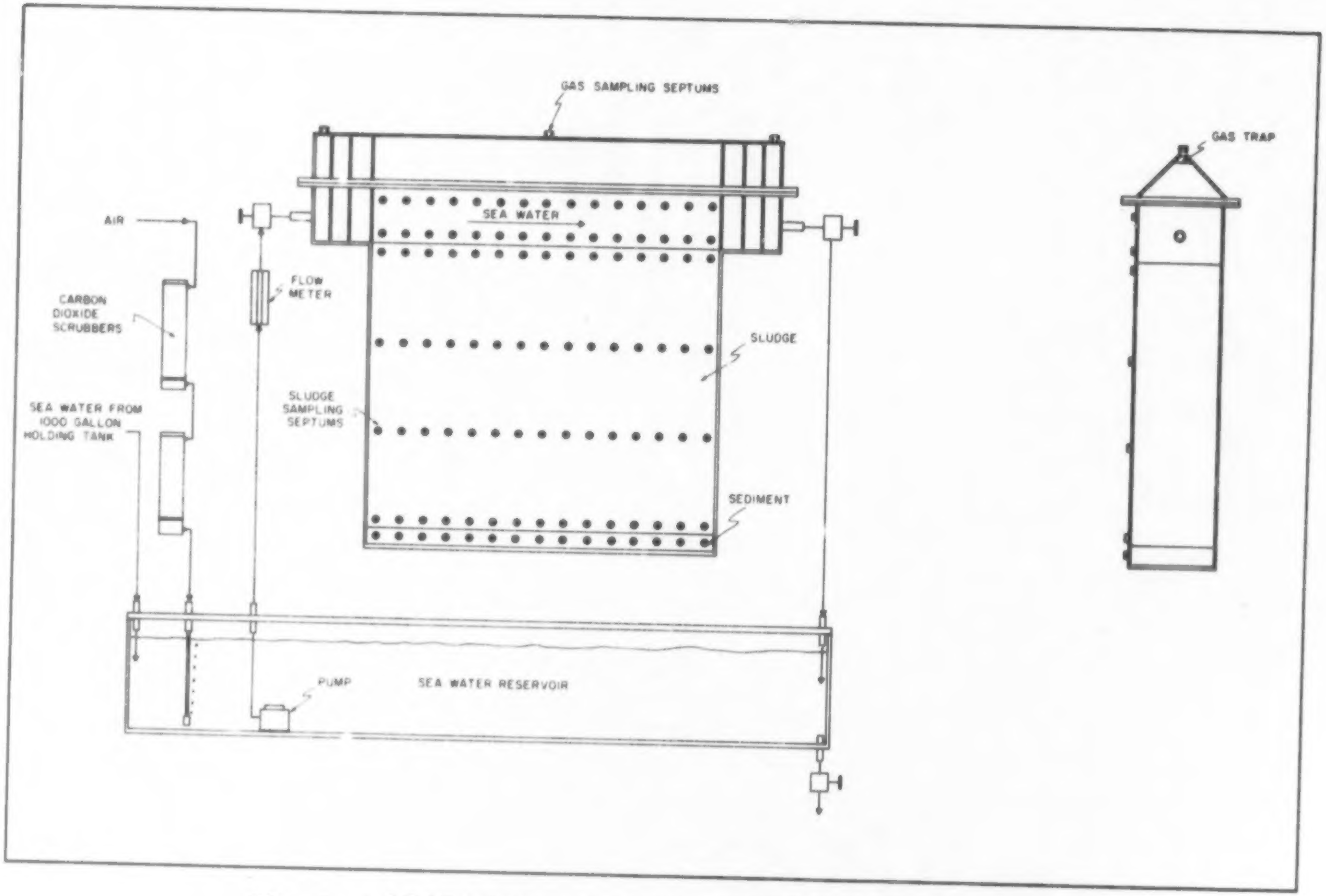
GAS SAMPLING SEPTUMS
GAS TRAP
SEA WATER
AIR
CARBON DIOXIDE SCRUBBERS
FLOW METER
SLUDGE
SLUDGE SAMPLING SEPTUMS
SEA WATER FROM 1000 GALLON HOLDING TANK
SEDIMENT
PUMP
SEA WATER RESERVOIR

Fig. 9 LABORATORY SLUDGE REACTOR - MOD II
(1 ATMOSPHERE)

CONCLUSIONS

Field and preliminary laboratory experiments have provided data on the decreases in sludge bed carbon concentrations and oxygen consumption rates. Further experiments are being devoted to better definition of initial carbon losses and documentation of carbon decreases in deeper beds. Measured signficant losses of organic carbon upon settling the sludge through a sea water column are being better documented because of possible ramifications regarding disposal procedures. A conceptual mathematical model of sludge bed dissolved, particulate and gaseous carbon fractions is being formulated in an effort to better understand the mechanisms and transfer rates for carbonaceous materials in stabilizing sea floor sludges.

Decreases in sludge bed total carbon content (grams of carbon per gram solid) amounted to 71 and 68% respectively for 81 and 68 days experiments at 1 atmosphere. Sludges subjected to pressures of 6.8 and 34 atmospheres had a carbon content loss of approximately 30% in 47 days. Reaction velocity constants (K) of 0.006 and 0.007 for the pressure experiments were lower than for experiments at 1 atmosphere where K ranged from 0.010 to 0.046.

The in-situ oxygen consumption rate of an aerobically digested sludge ranged from 3.74 grams O_2/m^2/day to 2.0 g-O_2/m^2/day after 65 hours. For an anaerobically digested sludge the initial rate of 2.51 g-O_2/m^2/day in 70 hours.

Laboratory measurements with anaerobically digested sludges revealed significantly different oxygen consumption rates depending on the sludge source. Sludge from a local treatment plant had an initial uptake rate of 2.72 g-O_2/m^2/day which decreased to 0.95 g-O_2/m^2/day after 53 days.

Sludge from a New York treatment plant required 0.72 g-O_2/m^2/day initially and 0.55 g-O_2/m^2/day after 20 days exposure to aerated seawater.

APPENDIX

Table 1. Carbon and Solids Determinations 1 Atm. Reactor, Test 1

1 ATMOSPHERE REACTOR TEST 1 (N.Y. STP SLUDGE)	T_o	T_o + 41 DAYS		T_o + 81 DAYS	
	ALL DEPTHS	SURFACE	2cm	SURFACE	2cm
TOTAL CARBON					
mg-c/l sample	12620	1992	5557	2110	5969
mg-c/g solid	329.5	48.0	97.0	46.5	94.0
TOTAL INORGANIC CARBON					
mg-c/l sample	234.1	115.7	128.5	26.5	96
mg-c/g solid	6.1	2.8	2.2	0.6	1.5
TOTAL ORGANIC CARBON*					
mg-c/l sample	12385.9	1876.3	5428.5	2083.5	5873
mg-c/g solid	323.4	45.2	94.8	45.9	92.5
*calculated					
TOTAL SOLIDS					
mg/l sample	38300	41471	57289	45418	63490
mg/g sample	39.8	40.9	56.6	45.0	62.5
TOTAL VOLATILE SOLIDS					
mg/l sample	27600	9140	17164	0566	14923
mg/g sample	28.7	9.0	17.0	10.5	14.7
% total solids	72.1	22.0	30.0	23.3	23.5
ASH (measured)					
mg/l sample	10701	32331	40125	34882	48567
mg/g sample	11.1	32.0	39.7	33.1	47.8
% total solids	27.9	78.0	70.0	76.8	76.5

NOTE: T_o sludge not settled through a sea water column before being placed in reactor (this test only).

Table 2 Carbon and Solids Determinations 1 Atm. Reactor, Test 2

1 ATMOSPHERE REACTOR TEST 2 (N.Y. STP SLUDGE)	T_o	T_o + 42	
	ALL DEPTHS	SURFACE	2cm
TOTAL CARBON			
mg-c/l sample	9584	2903	9440
mg-c/g solid	113.5	59.6	71.5
TOTAL INORGANIC CARBON			
mg-c/l sample	423.7	72.5	186.9
mg-c/g solid	5.0	1.5	1.4
TOTAL ORGANIC CARBON*			
mg-c/g sample	9160.3	2830.5	9253.1
mg-c/g solid	108.4	58.1	70.1
*calculated			
TOTAL SOLIDS			
mg/l sample	84468	48729	132072
mg/g sample	83	78.4	125
TOTAL VOLATILE SOLIDS			
mg/l sample	21980	10355	22794
mg/g sample	21.6	10.3	21.7
% total solids	26.0	21.3	17.3
ASH (measured)			
mg/l sample	62488	38383	109278
mg/g sample	61.4	38.1	103.8
% total solids	73.9	78.8	82.7

Table 3 Cárbon and Solids Determination, Pressure Vessel, 100PSI

PRESSURE VESSEL REACTOR 100PSI (N.Y. STP SLUDGE)	T_o	T_o + 47 DAYS	
	ALL DEPTHS	SURFACE	2 cm
TOTAL CARBON			
mg-c/l sample	8425	2486	9250
mg-c/g solid	126.0	55.4	96.1
TOTAL INORGANIC CARBON			
mg-c/l sample	180.5	89.6	168.6
mg-c/g solid	2.7	2.0	1.8
TOTAL ORGANIC CARBON*			
mg-c/l sample	8244.5	2396.4	9081
mg-c/g solid	123.3	53.4	94.4
*calculated			
TOTAL SOLIDS			
mg/l sample	66872	44843	96222
mg/g sample	66.2	44.2	93.5
TOTAL VOLATILE SOLIDS			
mg/l sample	22022	9988	24829
mg/g sample	21.8	9.8	24.1
% total solids	32.9	22.3	25.8
ASH (measured)			
mg/l sample	44850	34846	71393
mg/g sample	44.4	34.3	69.4
% total solids	67.1	77.7	74.2

Table 4 Carbon and Solids Determinations, Pressure Vessel, 500PSI

PRESSURE VESSEL 500PSI (N.Y. STP SLUDGE)	T_o	T_o + 47 DAYS	
	ALL DEPTHS	SURFACE	2cm
TOTAL CARBON			
mg-c/l sample	10110	2580	10969
mg-c/g solid	119.7	56.3	83.7
TOTAL INORGANIC CARBON			
mg-c/l sample	256.3	63.1	221.2
mg-c/g solid	3.0	1.4	1.69
TOTAL INORGANIC CARBON*			
mg-c/l sample	9853.7	2516.9	10747.8
mg-c/g solid	116.7	55.0	82.0
*calculated			
TOTAL SOLIDS			
mg/l sample	84468	45789	131047
mg/g sample	83	45.2	124.2
TOTAL VOLATILE SOLIDS			
mg/l sample	21980	8797	23988
mg/g sample	21.6	8.7	22.8
% total solids	26.0	19.2	18.3
ASH (measured)			
mg/l sample	62488	36992	107061
mg/g sample	61.4	36.6	101.7
% total solids	73.9	80.8	81.7

Table 5 Carbon and Solids Determinations, Glass Reactor, Test 1

SEMI-STATIC GLASS REACTOR TEST 1 (Corvallis STP Sludge)	T_o	T_o + 46 DAYS		T_o + 68 DAYS	
	ALL DEPTHS	SURFACE	2cm	SURFACE	2cm
TOTAL CARBON					
mg-c/l sample	11278	6715	11274	4897	10208
mg-c/g solid	169.6	94.7	66.1	84.0	54.5
TOTAL INORGANIC CARBON					
mg-c/l sample	805.7	N/M	N/M	50.1	204.8
mg-c/g solid	12.1			0.8	1.1
TOTAL ORGANIC CARBON*					
mg-c/l sample	10472	—	—	4846.9	10003
mg-c/g solid	157.5			83.2	53.4
*calculated					
TOTAL SOLIDS					
mg/l sample	66490	70890	170473	58259	187492
mg/g sample	68.7	71.2	167.0	58.9	173.8
TOTAL VOLATILE SOLIDS					
mg/l sample	28868	20393	31463	14824	31282
mg/g sample	29.8	20.5	30.8	15.0	29.0
% total solids	43.4	28.8	18.4	25.4	16.7
ASH (measured)					
mg/l sample	37625	50497	139010	43435	156210
mg/g sample	38.9	50.7	136.2	44.0	145.0
% total solids	56.6	71.2	81.6	74.6	83.3

N/M = not measured

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Bioassay Methods and Impact Evaluation of Ocean Disposal Sites

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Introduction

The use of ocean waters as final receivers of effluents and sludges (a liquid containing a certain percentage of solids) exists because of difficulty with land disposal due to (1) lack of adequate space for acceptance of large quantities of effluent and sludge, (2) possible adverse effects on the water table (eg. heavy metals introduced from industrial effluents and sludges, and nitrates and phosphates from both industrial and domestic effluents and sludges), (3) aesthetics, and (4) high land costs.

Coastal effluent and sludge discharges into ocean waters will probably expand in the future due to the effort to provide cleaner inland waters. Because the coastal concentration of the population is increasing, the volume of waste, both domestic and industrial, is increasing. In addition, a number of nuclear-powered generating plants are planned for the coastal region in order to utilize the large quantities of water needed for cooling. The resultant warm-water discharges will increase temperatures in marine waters.

The assimilative capacity of the oceans for wastes is not limitless. Dramatic changes, both biological and physical, occur over undefined areas around effluent discharges and disposal sites.

Ocean disposed effluents and solids, in quantity, result in increased nutrients and toxicants, have a biochemical and chemical oxygen demand, reduce directly or indirectly the dissolved oxygen, alter salinities, increase turbidity and temperatures, smother benthic organisms, and probably cause a number of synergistic and antagonistic reactions of undetermined consequences.

Nutrients, obtained from domestic and industrial effluents and sludge, such as nitrogen and phosphorus, serve as growth factors for algal plant life (Fig. 1). This may be beneficial if the type plant species and the additional organisms could be utilized as a food source by animals higher in the food chain. Chen and Orlob (1972) found increased numbers of an amphiurid sea star and amphipods in the vicinity of the San Diego outfall. Population maxima were found about two miles from the outfall. There may however be detrimental effects of increased nutrient loads in concentrated areas. Concentrations of nutrients may cause a bloom of organisms responsible for red tide. In addition, since all plants are not utilized by animals, there is a constant rain of dead, uneaten organisms onto the sea floor. This may, to an undetermined degree, increase the rate of eutrophication.

Toxins, including arsenic, cadmium, copper, cyanide, lead, nickel, mercury, and zinc, may inhibit or kill certain plants or animals. Toxins which have been diluted to low levels in the ocean may be accumulated within some organisms due to their tendency to concentrate certain toxic materials. Filter feeders, such as oysters and clams, are examples of such organisms (Table 1). Organisms higher in the food chain that feed on concentraters of toxins may suffer adverse physiological effects. For example, mercury discharged from a factory in Minamata, Japan, was diluted in a bay, but accumulated in edible fish, which resulted in a number of cases of mercury poisoning in humans.

The biochemical oxygen demand (B.O.D.) of effluents and sludge may make water unsuitable for certain animals and, concomitantly, may be indirectly responsible for introducing different organisms (eg. anaerobic bacteria and blue-green algae) to the discharge area. This situation could occur if the effluent consumed the available dissolved oxygen in the area about the discharge site. Depletion of dissolved oxygen to certain levels (eg. below 5.0 mg/l) can make the environment intolerable for some organisms such as certain fish species. In an intolerable environment the animals disappear or are killed. Complete depletion can result in noxious odors with the destruction of aesthetic values and elimination of both contact and noncontact recreation activities. In addition, the new conditions may result in the establishment of undesireable organisms (eg. anaerobic bacteria that produce noxious hydrogen sulfide gas).

Salinity changes, due to the introduction of fresh water, occur in the outfall or dumping areas. Effluent and sludge are usually warmer and less saline than seawater and rise to form plumes when introduced through an outfall pipe. The currents disperse and dilute this freshwater addition, but for some area above and around the dispersal point the ocean salinity is reduced. Marine animals and plants that can tolerate only narrow salinity changes would disappear, or attempt to avoid these diluted waters.

Thermal discharges may have adverse impacts on the marine environment in several ways. Heat affects the physical properties of water (eg. density, viscosity, and reduces the solubility of oxygen in water). Heat speeds the rate of chemical reactions, and of biological reactions. Increased temperatures may reduce the number of species in the community and stimulate excessive populations of individual species. Increased temperatures may result in synergistic reactions (eg. increased temperatures may heighten the toxic effects of some materials). Finally, sudden temperature changes may have an immediate adverse effect on fish and other biota. Also, if the dispersal current carries a thermal plume across an access point (eg. a bay, or river mouth) movements of migratory species may be prevented.

Increases in turbidity due to sludge dumping reduces incident light. This results in reduced photosynthetic activity and concurrently, make less food available for those animals that feed on the primary producers.

Studies that have been made of specific sludge dumping grounds indicate that sludge disposal may (Barber and Krieger, 1970; Grigg and Kiwala, 1970; National Marine Fisheries Service, 1972) or may not (Shelton, 1971; Davey, 1972) adversely effect marine or estuarine habitats. The variations reported are probably due to the difference in the type of, or the constituents within, sludge disposed; to the behavior of currents in the disposal site; to the quantity of disposed materials and the size of the dispersal site; and to combinations of these and other oceanographic factors, including water chemistry. A number of interactions can occur that would alter the toxicity of the effluent or sludge. For example, high phosphate additions might render lead less toxic due to precipitation (Shelly and Steucek, 1973). Lee (1973) has indicated that the toxicity of a pollutant may depend on the form in which the element is found and that may depend on such factors as the pH and redox potential of the receiving water.

Assay Techniques

Any bioassay or impact evaluation program, when applied to a particular outfall, should consider each of the effects previously mentioned.

A number of both laboratory and field procedures are available today to determine the effects of ocean disposed materials on plants and animals. However, to perform bioassays that require the use of marine waters, some modifications of the standard techniques used for freshwater bioassays (Standard Methods, 1971) are necessary. For example, organisms with some degree of salt tolerance must be employed. In addition, the apparatus utilized in the performance of laboratory tests should not be constructed of metals or toxic substances when these parts would come into contact with salt water. The use of receiving waters, ie. 'natural' waters, for determining effects of effluents or sludges, while preferred, may pose problems. Diluting waters of unknown quality may, at different times, have possible additive, antagonistic, or synergistic effects on the effluent or sludge to be examined. Further, utilization of receiving waters may, at any undefined time, result in acute toxic effects due, not only to the discharged material, but also to undetermined toxicants in the diluting assay waters. Since receiving waters may be of questionable quality, and may be difficult to obtain, waters of a constant quality with regard to their average dissolved mineral content could be prepared. This would provide waters with uniform composition, and avoid background toxic materials. This would result in an assessment of the toxicity of the effluent as if discharged into a non-stressed body of water.

Lee (1973) indicated that the oxidation state, solubility, complexation, ionic strength, type and amount of solids, salt ratios and concentrations, and organic content of waters cause changes in the environment of a bioassay that may affect the results of the test. He indicates that cooperation between chemists and biologists is necessary in order to produce meaningful results for bioassays.

Types of Bioassays

Two types of bioassays may be considered;

A. Static bioassays.

A static bioassay is one in which the test organisms are maintained in vessels containing solutions of various dilutions of the toxic material. The concentration at which half the organisms remain after a predetermined period (from minutes to 24, 48, or 96 hours) is known as the tolerance level median (TL_m). This gives an estimate of the acute toxicity of the material. Sublethal toxicity may be determined in a similar way except that the effect of the toxin on some physiological activity, such as respiration, is determined. The static bioassay is a relatively simple technique, and may, in some cases, be used in the field.

This test has some shortcomings. Some, or all, of the toxicity of the test solution may be lost during the assay due to absorption, adsorption or volatilization of the toxin. In addition, the test organisms may excrete wastes that change the character of the test water.

B. Flow-through bioassays.

A flow-through bioassay differs from the static bioassay in that the test solution is allowed to flow into and out of the vessel containing the test organisms, maintaining a constant chemical environment. For this reason, the flow-through system is the preferred method for longer-term assays.

The organisms selected for either the static or flow-through bioassay should be: (1) readily available, (2) sensitive, to some degree, to the toxin or toxins of interest, (3) relatively easy to handle and observe, (4) an organism that is found part or all of the time near the area affected by the effluent or sludge. It might also be desireable to use a "standard" organism, such as the brine shrimp, Artemia salina (Fig. 2), for all ocean outfalls regardless of location, so that comparisons among outfalls could be made. It would be advantageous to use more than one organism, including one or more standards, and one or more locals. Temperature and salinity at the time of capture should be used as the test temperature and salinity when using field organisms to evaluate toxicity. This reduces the problems associated with longer term acclimation of the organism, and of possible stresses arising from incomplete acclimation.

Certain animals are not generally considered as acceptable test organisms because of a considerable tolerance, relative to other organisms, to adverse conditions. The mummichog, Fundulus heteroclitis, is such an organism. However, because of its abundance in certain waters, for example, Newark Bay, New Jersey, any assay performed with this water certainly should include this species, since it is probably the main food source of carnivorous fishes in the bay.

A number of other invertebrates have been used as bioassay organisms, including hydrozoans, amphipods, isopods, shrimp, crabs, clams, oysters, and snails.

Woelke (1972), has detailed a bioassay procedure using embryos of a commercially valuable marine organism, the Pacific oyster (Crassostrea gigas). This bioassay can be applied to marine waters at any time of the year with more ease and reproducibility than certain other bioassays. The test determines the number of abnormal 48-hour Pacific oyster embryos when exposed to a level of stress, or toxicant. The principle measurement employed has been the percentage of abnormal larvae found after 48 hours exposure of fertilized oyster eggs to the variable under consideration. Failure to develop as normal larvae results in their death, breaking the life cycle of the oyster at the reproductive stage. He recommends a laboratory with a flowing seawater system that is pollution free.

To determine the effects of ocean disposal on adult clams or oysters, a long-term flow assay could be performed to determine the effect of disposal waters on gaping times. This could be accomplished by pasting the clams or oysters to a basin, attaching one valve to a kymograph apparatus, and exposing one group of test organisms to disposal waters and a second to control waters (either 'natural' or synthetic) (Dorfman, unpublished data).

Problems with static and flow-system bioassays include; (1) laboratory conditions that do not wholly emulate field conditions (eg. the choice of synthetic seawater or filtered local seawater). Unlike the typical single stress laboratory test, the natural environment is liable to create several different stresses simultaneously. (2) the difficulty in duplicating effluent or sludge composition, which may vary hourly, or daily (this could be minimized by performing repeated bioassays, and by using grab samples of the materials to be disposed. (3) the virtual impossibility of assaying the effect of the disposed material on all of the organisms that live for part of all of their lives in the disposal area. (4) the questionable extrapolation of data, obtained for one or more species to other species that may or may not react in a similar fashion (ie. with the same response).

The ideal situation in determining the toxicity of an effluent or sludge into a discharge area includes the determination of its effect on the most susceptible organism naturally occurring in the area, that is the"weakest link" in the food chain. Since the response of the weakest organism, or the most susceptible organism, to the effluent may not be determined because the weakest organism may not have been selected as the test organism, an application factor could be applied to the data suggested by the TL_m, or from data obtained from sub-acute responses during flow-through bioassays. The TL_m as an acceptable number for allowable dilution is not in itself satisfactory, since the death of 50 percent of the test organisms is not an acceptable rate of mortality for anything other than the test itself. In addition, the TL_m offers no clue as to the sub-acute or chronic effect of the effluent on the organism. Sub-acute effects include (1) reduced fertility, (2) difficulty in movement, (3) certain physiological disorders, (4) greater susceptibility of the organism to disease or predation, (5) the death of the organism after five or more days of exposure.

Application factors, applied to the obtained data, would quantify allowable concentrations and dilutions that would permit the survival of the most susceptible organism. Mount and Stephan (1967) have suggested a technique for determining permissable concentrations of toxicants based upon quantitative experimental data.

They proposed the use of a "laboratory fish production index" (LFPI) as the measure of effect to furnish a first approximation as to whether an environment is unacceptable to fish. For this index, exposure data would be obtained for at least one generation and would reflect effects on growth, reproduction, spawning behavior, viability of eggs, and growth of the fry. The maximum acceptable toxicant concentrations would be established on the basis of chronic exposure using LFPI as the means of effect. They indicated that the application factors derived for one species and one kind of water have not been shown to be applicable to other species and waters. The duration of their tests were nine months. Their application factors, for freshwater minnows, obtained after determining 96 hour TL_ms, ranged from 1/15 to 1/45 of the 96 hour TL_m.

Sub-lethal tests

A number of sub-lethal tests can be performed. Biochemical studies-These include studies on enzyme induction and inhibition. They provide information on the nature of the toxic action. They are useful in predicting side effects of toxins. The information is not however readily obtained. Pathological studies-Preparations of tissue and organ sections from animals exposed to lethal and sub-lethal levels of toxic substances can give an indication of the site of toxic action particularly if the damage is severe. The difficulty lies in the size of the organism being studied. Included in pathological studies would be hematological changes from the normal (eg. hematocrits), and electrophoretic studies of serum and plasma proteins, which might be indicative of stressful conditions due to ocean disposal (Fig. 3).

Physiological studies-These studies would determine the effects of wastes, including thermal discharges, on osmoregulation, changes in heart and respiration rates, etc.

Growth studies-These studies would include, with animals, studies of longer durations (eg. life cycles of fishes could be examined after exposure to sublethal quantities of wastes as constant dosages or as slug loads).

Behavior studies-Avoidance studies could be performed to determine the reactions of organisms to gradients of pollutants (eg. oxygen gradients, temperature gradients, heavy metal gradients). In addition, swimming speeds, and general activity of the organism could be observed.

Photosynthesis studies-The photosynthetic activity of planktonic algae can be studied with mixed, or single cultures (eg. *Massartia rotundatum*, *Navicula sp.*, *Chloromonas sp.*, *Amphora sp.*) (Mountford et al.,1973, unpublished data; Gesamp, 1972) incubated in bottles, or in the field, using carbon 14 techniques. These are tests designed to run for short terms (minutes or hours). The data generated can be indicative of short-term damaging effects, and of changing phytoplankton species composition.

Taste tests-The flesh of fish can acquire a taste from organic compounds in the water, therefore taste tests can be made to determine the tainting of fish flesh at concentrations below the lethal level.

Other laboratory procedures are available to assess sublthal or chronic effects. The determination of hematological characteristics for dertain species could be determined as base-line data and periodic examination of these species inhabiting the disposal area could be made to determine if subtle changes had occurred. The coughing response of fish has recently been suggested as a method of determining unfavorable environments (Newsweek,1974). Apparently some fishes have a coughing reflex when exposed to low concentrations of metals and pesticides.

Field Methods

Field observations, although time consuming and costly, provide the only direct evidence of the effect of ocean waste disposal on the local flora and fauna. Field methods should include base-line data surveys of the general area of the disposal site. This would include surveys of both flora and fauna, including benthic organisms. Chemical features of the surrounding waters would also be examined.

The effect of a particular outfall or sludge dumping site in the sea can be assessed by field sampling and determination of the distribution of benthic organisms (Reish, 1960; National Marine Fisheries Service, 1972). Many species may be used as indicators of pollution by their absence or presence in the area of the disposal site (Table 2).

The variety of organisms, feferred to as species diversity, has been found to be a useful indicator of marine pollution (Bechtel and Copeland, 1970; Copeland and Bechtel, 1971; Norris, et.al., 1973). In general, the more polluted, highly stressed, environments are characterized by low species diversity, and cleaner waters by a greater diversity of species. Field work is an important means of determining the effects of various point source discharges as long as they are not in close enough proximity to each other to produce overlapping effects.

Field observations are not without their problems however. For example, some fish species are migratory by nature, especially those that are pelagic. Other species, both plant and animal, have typical seasonal cycles. Therefore, the presence, or absence, of one or more species may not be due to ocean disposal effects, but due to the cyclical appearance or disappearance of that organism.

A multidisciplinary approach appears to be the most satisfactory one. Determination of the disposal site far enough in advance of its use to allow adequate base-line data surveys, including chemical, physical, and biological parameters, should be made. Subsequent field and bioassay monitoring, on a programed basis, would then be utilized after waste disposal began. In addition, no single test organism can provide the necessary data to determine the impact evaluation of a disposal site. Extrapolation of the responses of one species, or of several species, might not include the response of the most sensitive species if that organism was not included among the test species. A multidisciplinary approach with constant monitoring would possibly provide enough data to assess the impact on single species and, more importantly, on the food web. Additional safeguards, as promulgated by bioassays and impact evaluation, could then be made at the effluent and waste disposal sources to reduce possible toxicants to levels manageable by marine organisms.

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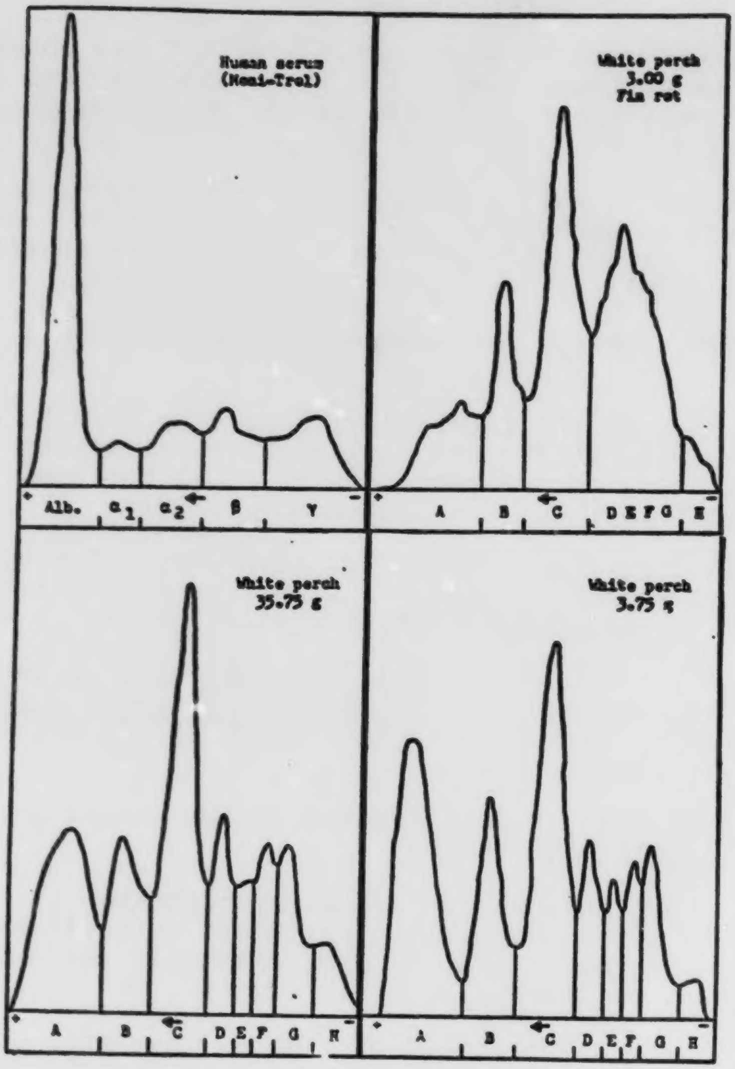
Human serum
(Moni-Trol)
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Alb.
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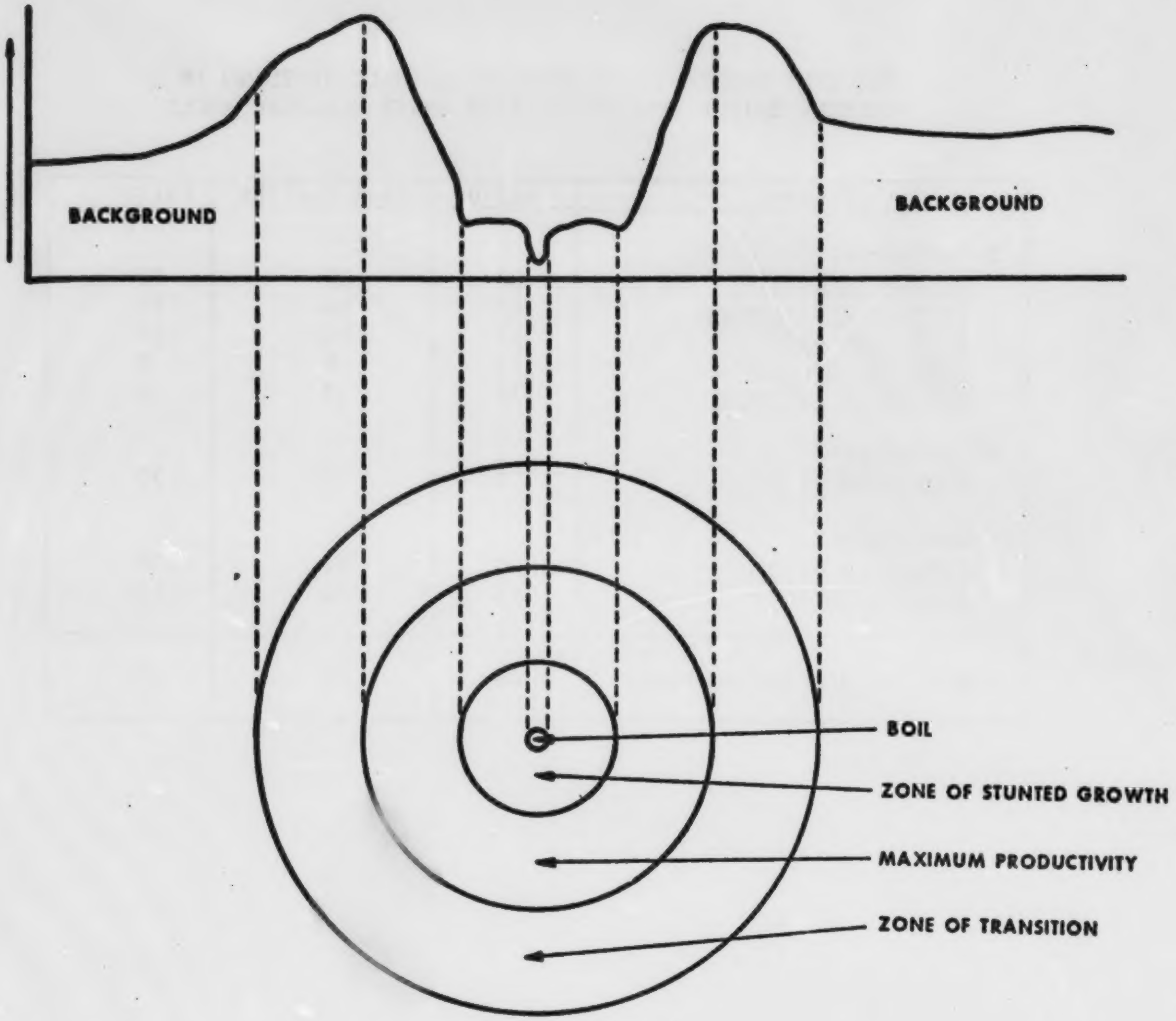

BACKGROUND
BACKGROUND
BOIL
ZONE OF STUNTED GROWTH
MAXIMUM PRODUCTIVITY
ZONE OF TRANSITION

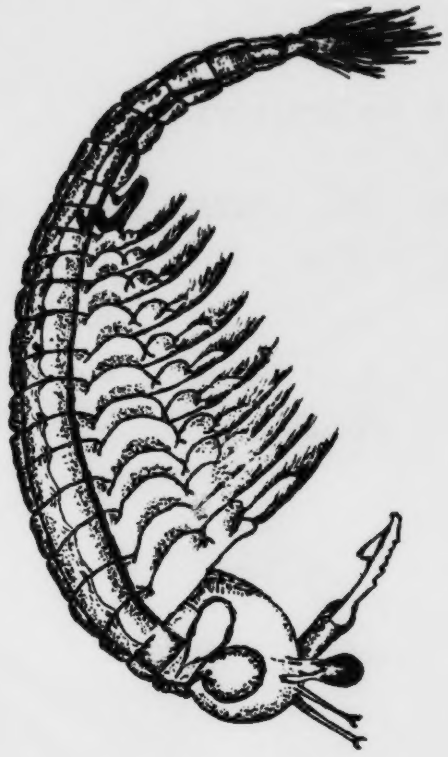
PRODUCTIVITY PATTERN IN AREA OF OCEAN OUTFALL

PER CENT OCCURENCE OF DOMINANT ANIMALS SETTLING IN SEDIMENT BOTTLE COLLECTORS FROM THREE ECOLOGIC AREAS

Species	Healthy	Semi-Healthy	Polluted
Polychaetes:			
Capitella capitata	88	91	78
Podarke pugettensis	62	88	59
Dorvillea articulata	52	79	65
Armandia bioculata	73	9	2
Halosydna johnsoni	31	13	2
Oligochaetes:			
Tubificid	2	7	30
Crustaceans:			
Corophium acherusicum	96	64	20
Epinebalia sp.	13	78	63
Number of Bottles Analyzed	48	67	49

TRACE METAL CONCENTRATIONS (ppm)

Type Sample	Zinc	Copper	Cadium	Chromium	Lead
Oysters	495.0	27.8	1.12	Trace	no[b]
Sediment	61.0	2.1	0.31	2.8	3.7
Water	0.012	0.007	0.01	0.004	0.008
Interium Standard	1500.0	100.0	0.2	—	0.2



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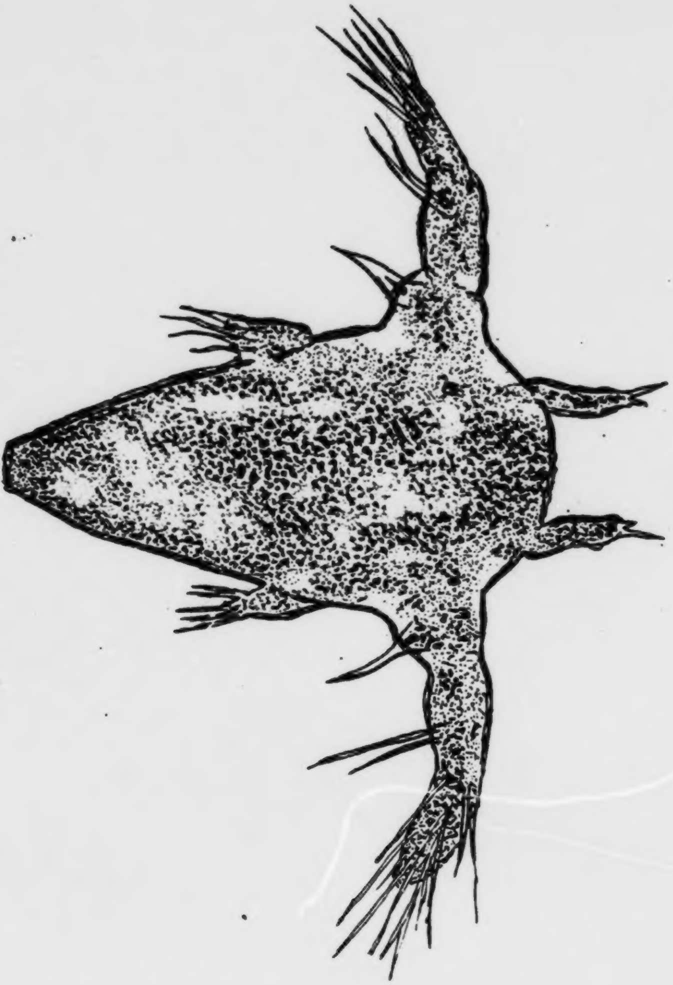

NAUPLII LARVAE
COWELL

EVLUATING THE IMPACT OF SLUDGE DISCHARGE TO SANTA MONICA BAY, CALIFORNIA

BY

Floyd K. Mitchell*

Municipal wastewater from the City of Los Angeles is treated at Hyperion Treatment Plant and discharged to Santa Monica Bay through two separate submarine outfalls. One outfall consists of a 5 mile (8 km) long, 12 foot (3.66 m) diameter concrete pipeline and two 4,000 feet (1220 m) long Y diffuser legs through which approximately 335 mgd (1,270,000 cu m/day) of combined primary and secondary effluent is discharged at a depth of 200 feet (61 m). The other outfall is a 7 mile (11.3 km) steel pipeline, 22 inches (56 cm) in diameter, through which about 4.8 mgd (18,150 cu m/day) of combined digested sludge and secondary effluent are discharged at the head end of a submarine canyon at a depth of 320 feet (98 m). Both outfalls have been in operation since 1960. Figure 1 shows the location of these two outfall pipelines in Santa Monica Bay. As is evident from this figure, Santa Monica Bay is an open bay and the discharges represent essentially open ocean waste disposal. The numbered points in Figure 1 are water and sediment sampling and quality monitoring stations.

A flow diagram of the Hyperion Treatment Plant is presented in Figure 2. Flow rates (in mgd) are noted in parentheses. The entire flow of 340 mgd (1,290,000 cu m/day) receives primary treatment and approximately 100 mgd (378,500 cu m/day) of the primary effluent receives activated sludge secondary treatment. The raw sludge produced in the primary treatment amounts to 1.3 mgd (4,920 cu m/day) with an average total solids content of 6.4 percent. This primary sludge is anaerobically digested in 18 digesters which provide for a mean detention time of 25 to 30 days. The digested sludge is screened to remove large particulate matter and then diluted about 3:1 with secondary effluent prior to being pumped 7 miles (11.3 km) out to sea. The sludge screenings are hauled by truck to landfill.

The results of quantitative physical and chemical analyses of the two separate effluents are listed in Table 1. Although the flow of the sludge

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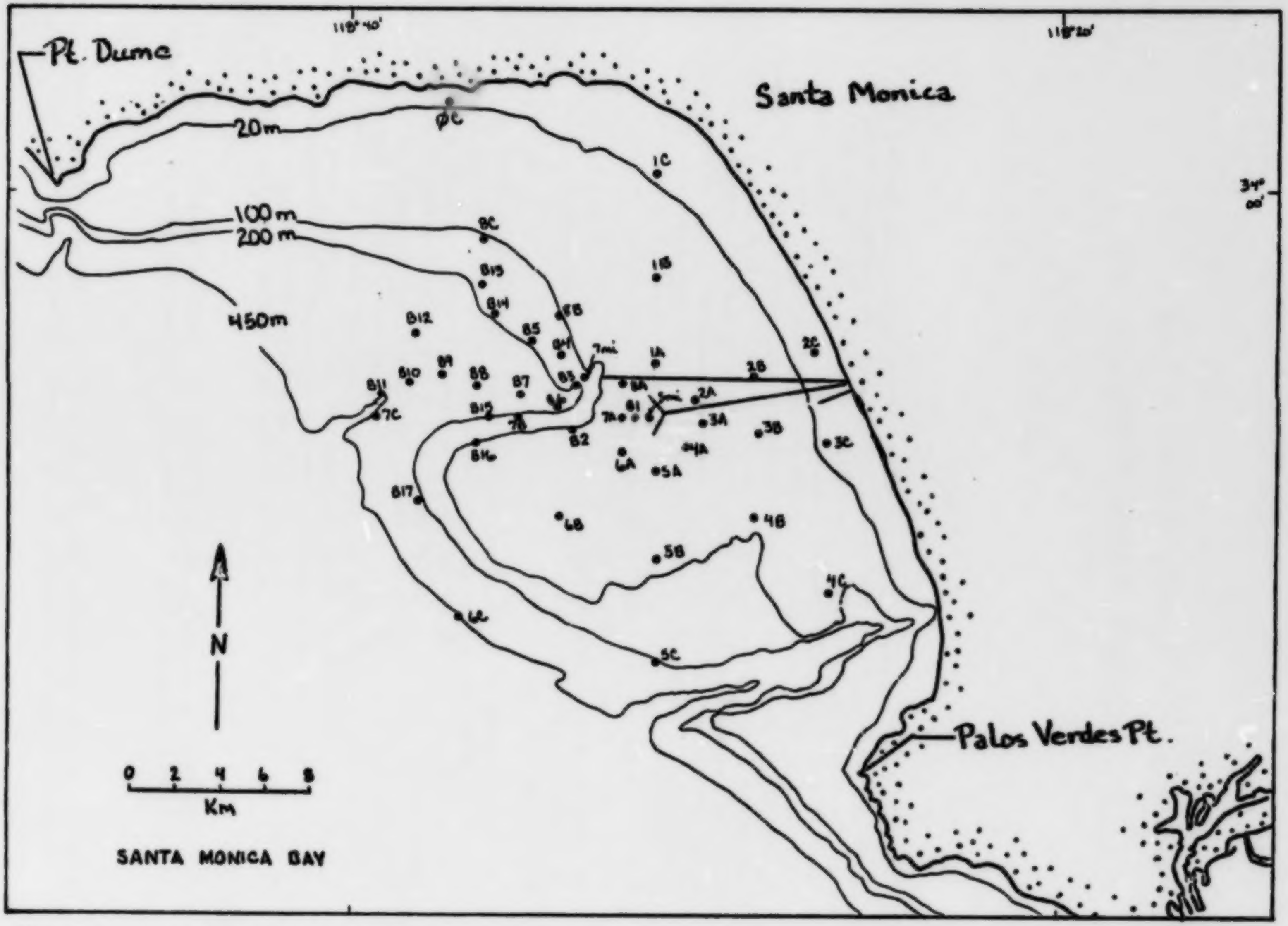
118°40'
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Pt. Dume
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20m
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SANTA MONICA BAY

Figure 1 Hyperion Outfall Locations

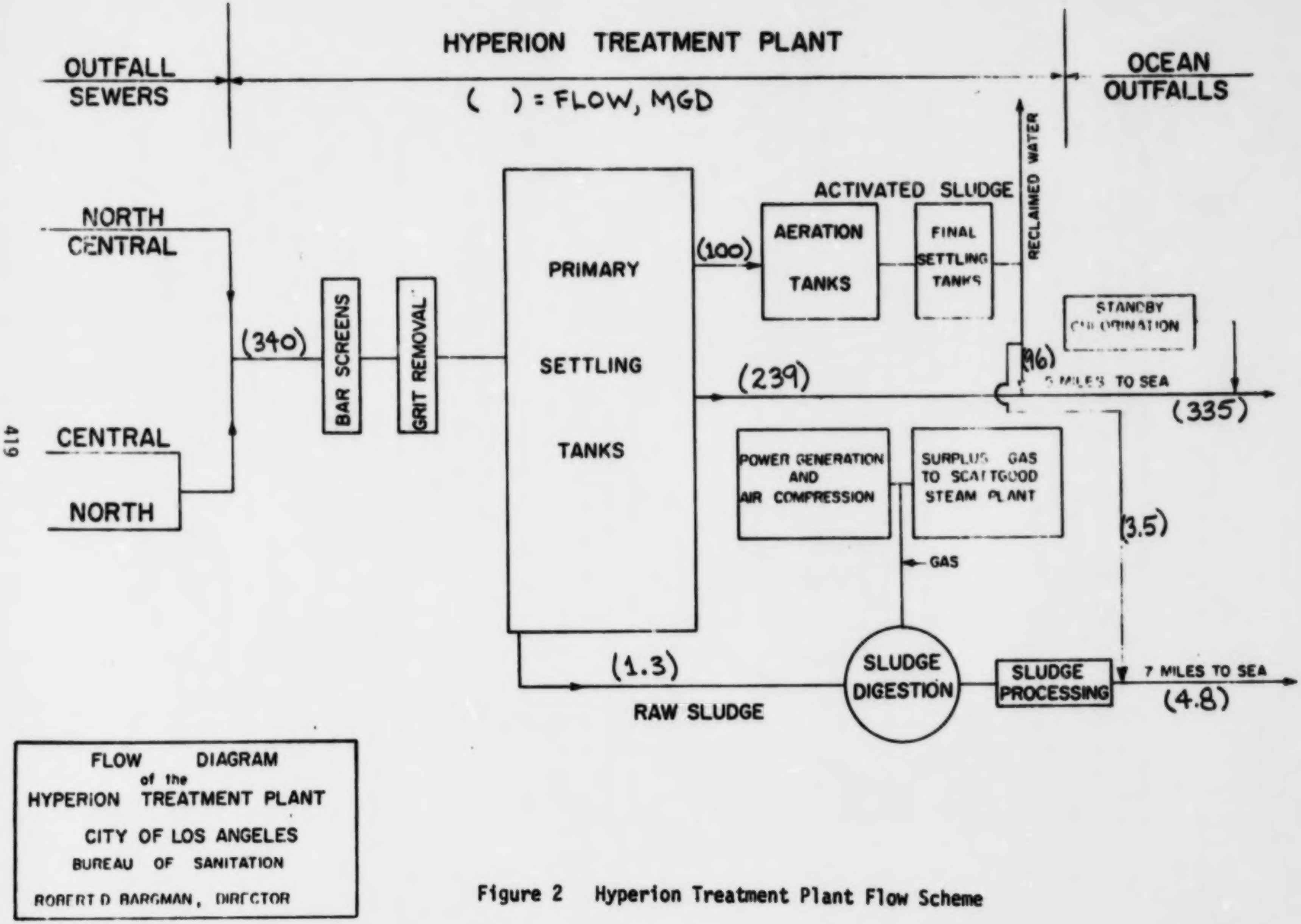

HYPERION TREATMENT PLANT
OUTFALL SEWERS
OCEAN OUTFALLS
( ) = FLOW, MGD
NORTH CENTRAL
CENTRAL
NORTH
(340)
BAR SCREENS
GRIT REMOVAL
PRIMARY SETTLING TANKS
(100)
ACTIVATED SLUDGE
AERATION TANKS
FINAL SETTLING TANKS
RECLAIMED WATER
STANDBY CHLORINATION
(96)
(239)
5 MILES TO SEA
(335)
POWER GENERATION AND AIR COMPRESSION
SURPLUS GAS TO SCATTGOOD STEAM PLANT
GAS
(3.5)
(1.3)
RAW SLUDGE
SLUDGE DIGESTION
SLUDGE PROCESSING
7 MILES TO SEA
(4.8)
FLOW DIAGRAM of the HYPERION TREATMENT PLANT
CITY OF LOS ANGELES
BUREAU OF SANITATION
ROBERT D BARGMAN, DIRECTOR

Figure 2 Hyperion Treatment Plant Flow Scheme

effluent is only 1.4 percent of the five mile effluent flow, the mass emission rates (MER) from the two outfalls are quite similar for most of the measured contituents. This comparison establishes the sludge discharge as a major contributor of waste material to the bay, and likely a more significant contributor in terms of effects due to the higher concentrations involved.

Sludge Characteristics

One phase of an assessment of the environment effects of any waste disposal practice must be a detailed characterization of the waste material being discharged. The phsical and chemical properties of four types of Hyperion Treatment Plant sludges are reported in Table 2. Twenty-six samples of each sludge type were collected and analyzed over a six week period at the end of 1972. The digestion of the primary sludge results in significant reductions of total solids, volatile matter, chemical oxygen demand, and grease without affecting total nitrogen and total phosphorous concentrations of the sludge. Heavy metals in municipal sludge is a subject which is of increasing concern, primarily because of the high concentration usually reported and the general lack of knowledge and accompanying apprehension regarding the ultimate fate and effects of these metals in a sludge disposal scheme. Table 3 presents some information on the heavy metals content of three Hyperion Plant sludges. The metals concentrations are divided into dissolved and suspended categories and the numbers in parentheses are estimates of the suspended metals content of the sludges in terms of mg/dry kg. The digested sludge values represent a combination of the mesophilic and thermophilic digested sludges metals content. For each of the six metals and all three sludge types, the metals are predominantly associated with the sludge particulate matter. It may be of interest to note that the return activated sludge solids metals content (mg/dry kg) are greater than the primary sludge solids in every case and greater than the digested sludge solids for four of the six metals reported.

Although the sludge characteristics reported here are useful in assessing potential effects of proposed disposal schemes and some expected effects of existing disposal practices, a knowledge of other sludge properties is desireable. Some of the more obvious properties of interest include biodegradability and degradation rates, toxicity, enrichment potential, metals mobilization potential and rates, and particle size distributions. It is only with information of this type along with the traditional physical and chemical composition parameters normally reported that a scientific assessment of potential and existing environmental effects of various sludge disposal practices and proposals can be seriously attempted.

Table 1

HYPERION TREATMENT PLANT WASTE DISCHARGES
1973 ANNUAL AVERAGE CONCENTRATIONS AND MASS EMISSION RATES

CONSTITUENT	FIVE MILE EFFLUENT (335 mgd)		SEVEN MILE EFFLUENT (4.8 mgd)	
	Concentration mg/1	M.E.R. M Ton/Yr	Concentration mg/1	M.E.R. M Ton/Yr
SUSPENDED SOLIDS	81	38,100	7,400	49,300
VOLATILE SUSP. SOLIDS	63	29,600	4,500	30,000
B.O.D.	98	46,100	-	-
C.O.D.	205	96,500	7,800*	51,900*
OIL AND GREASE	18	8,500	922	6,140
TOTAL NITROGEN	18.4	8,650	540*	3,600*
TOTAL PHOSPHORUS	6.3	2,970	145*	965*
TOTAL CHLORINATED HYDROCARBONS	.003	1.4	.033	.22
HEAVY METALS				
MERCURY	.0026	1.22	.14	.93
COPPER	.14	66	13.6	90.5
NICKEL	.17	80	3.6	24
ZINC	.25	118	27	180
CADMIUM	.02	94	.98	6.5
LEAD	.04	18.8	1.57	10.5
CHROMIUM	.29	137	18.2	121
SILVER	.05	23.6	.8	5.3

*Estimates, not annual averages

Table 2
HYPERION TREATMENT PLANT SLUDGE CHARACTERISTICS*

MEASURED CHARACTERISTIC		PRIMARY SLUDGE	DIGESTED SLUDGE		RETURN ACTIVATED SLUDGE
			Mesophilic	Thermophilic	
TOTAL SOLIDS, %	Range	4.25-8.90	1.15-2.98	1.67-3.64	0.44-0.89
	Mean	6.40	2.12	2.35	0.69
	Mode	6.35	2.36	2.47	0.69
VOLATILE SOLIDS, %TS	Mean	80.2	59.9	62.0	73.1
	Mode	81.0	61.5	62.9	75.0
CHEMICAL OXYGEN DEMAND mg/l	Mean	61,267	24,195	33,420	7,106
	Mode	47,900	24,270	38,000	6,820
TOTAL KJELDAHL NITROGEN, mg/l	Mean	2,066	2,000	1,992	541
	Mode	2,418	2,113	2,028	501
ORGANIC NITROGEN, mg/l	Mean	1,874	870	697	512
	Mode	2,320	845	720	504
TOTAL PHOSPHATE,mg/l	Mean	1,795	1,570	1,604	1,148
	Mode	2,000	1,596	1,665	1,174
GREASE, mg/l	Mean	11,080	1,981	2,820	400
	Mode	11,436	2,360	2,648	345
ALKALINITY mg/l	Mean	1,603	5,524	6,190	430
	Mode	1,420	5,500	5,750	440

*Based upon analyses of 26 samples of each type of sludge collected during the period 11/30/72 to 1/12/73

Analyses by Hyperion laboratory

Table 3
HEAVY METAL CONCENTRATIONS IN HYPERION SLUDGES

METAL	METAL CONCENTRATION, mg/l and (mg/dry kg)* PRIMARY SLUDGE Dissolved	Suspended	DIGESTED SLUDGE Dissolved	Suspended	RET. ACT. SLUDGE Dissolved	Suspended
CADMIUM	.02	2.48 (38.8)	.02	2.88 (123)	.03	.77 (131)
NICKEL	.67	10.83 (169)	.41	10.1 (427)	.38	1.42 (242)
LEAD	-	8.0** (125)	-	2.85** (324)	.25	2.25 (384)
ZINC	1.55	119.5 (1,880)	.13	72.55 (3,120)	.25	12.75 (2,180)
COPPER	11.0	52.0 (812)	6.0	34.4 (1,460)	.5	11.5 (1,960)
CHROMIUM	.09	45.2 (707)	.10	51.4 (2,185)	.07	16.0 (2,730)

* Estimated from mean solids content of sludges (Table 2), not reported in original data

**Total Content

Analyses by Hyperion Laboratory

Physical and Chemical Effects

The most obvious effect of the discharge of sludge to the ocean is an alteration of the physical and chemical characteristics of the marine sediments in the vicinity of the discharge site as a result of the sludge particulate matter settling to the bottom. Sampling and analysis of Santa Monica Bay and Santa Monica Canyon bottom sediments for heavy metals and chlorinated hydrocarbons content were conducted in 1970 and 1971. Sediment sampling stations were as shown in Figures 3 and 4. Some results of the analyses are presented in Figures 5-11.

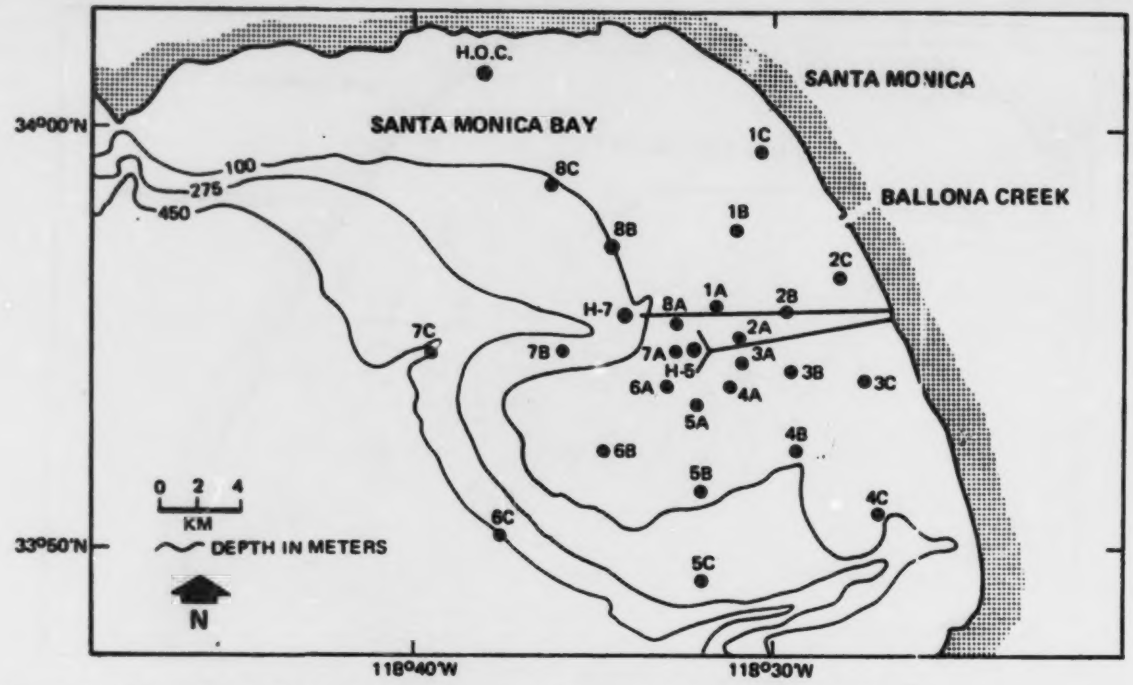
Figures 5 and 6 show the distribution of lead and copper concentrations found in the surface layer of the samples collected. Assuming a natural background concentration of 8 mg/dry kg for lead and 30 mg/dry kg for copper (SMC sample, Fig. 7), it is seen that Figure 5 indicates an enrichment factor for lead varying from 2 to 5 times background levels over a large portion of the bay area and as high as 31 times background at station B3, closest to the sludge outfall. Similarly, Figure 6 shows enrichment factors for copper of 1 to 3 over a wide area and 17 times background at station B3. The distribution of other metals in the surface sediments is similar to those of lead and copper. The highest concentrations found (station B3) range from 1/3 to 1/5 of the metals concentrations reported for the discharged sludge particulates.

Three box core samples (B3, B6 and B8) from the Santa Monica Canyon were analyzed for metals concentrations with respect to depth. Some results of these analyses are presented in Figure 7. The plots of copper, mercury, and lead concentrations versus depth reveal a similarity between the depth profiles for B6 and B8, and suggests that such profiles might be used to determine the depth of deposit of significant quantities of sludge particulates. The relatively constant depth concentrations at station B3 indicates that this sample was not as deep as the sludge particulates. Other metals in these cores exhibit similar depth profiles.

Chlorinated hydrocarbons in the surface sediments at the same locations present a picture similar to that of the metals (Figure 8-11). Total DDT and total PCB concentrations are higher in the canyon than in the shallower areas of the bay and are highest by far at locations closest to the sludge outfall.

Biological Effects

The physical and chemical changes in sediment characteristics resulting from the deposition of wastewater and sludge particulates on the ocean bottom would be expected to have a quantitative effect on the benthic fauna. The most commonly used parameters for assessing the health of a biological community are total biomass per unit area, species composition, species diversity, and if possible, organism abnormalities. Data gathered to date indicates that the discharge of wastewater and sludge to Santa Monica Bay results in an alteration of the polychaetes, high proportions of capitellid species are commonly regarded as being indicative of environmental stresses such as reduced salinity, low dissolved


H.O.C.
SANTA MONICA
SANTA MONICA BAY
34°00'N
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275
450
8C
BALLONA CREEK
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2C
1A
2B
8A
H-7
2A
7C
7B
7A
3A
H-5
3B
3C
6A
4A
5A
4B
6B
0 2 4
KM
5B
4C
6C
33°50'N
DEPTH IN METERS
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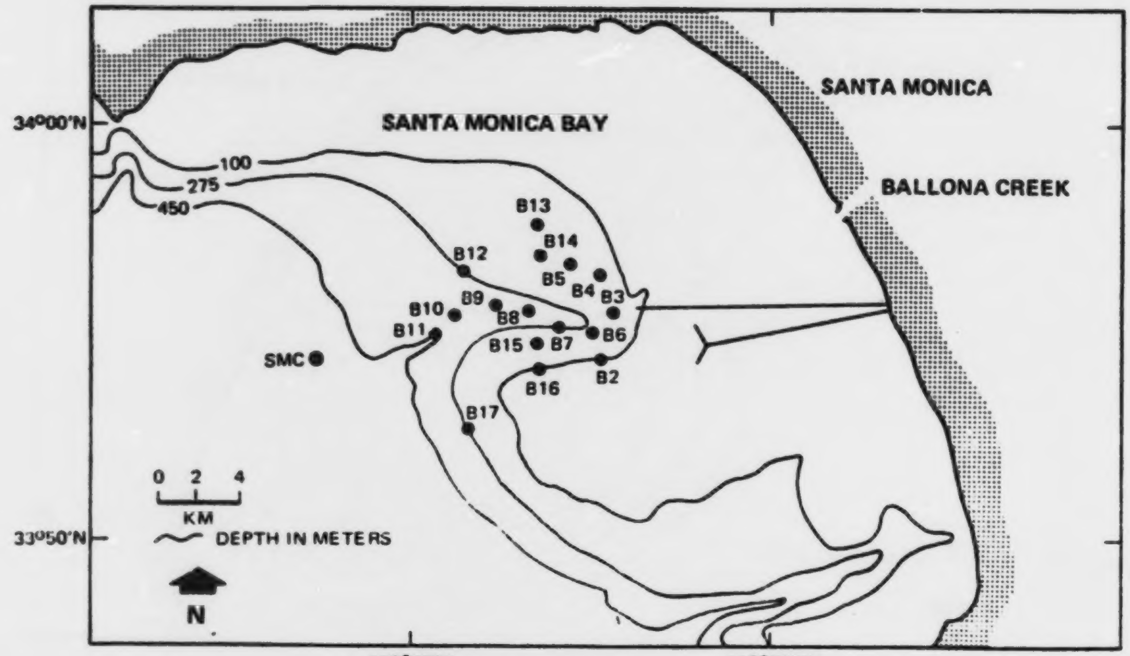
N
118°40'W
118°30'W

Figure 3 Phleger Core Collection Locations Around the Hyperion Outfall System, June 1970.


SANTA MONICA
34°00'N
SANTA MONICA BAY
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275
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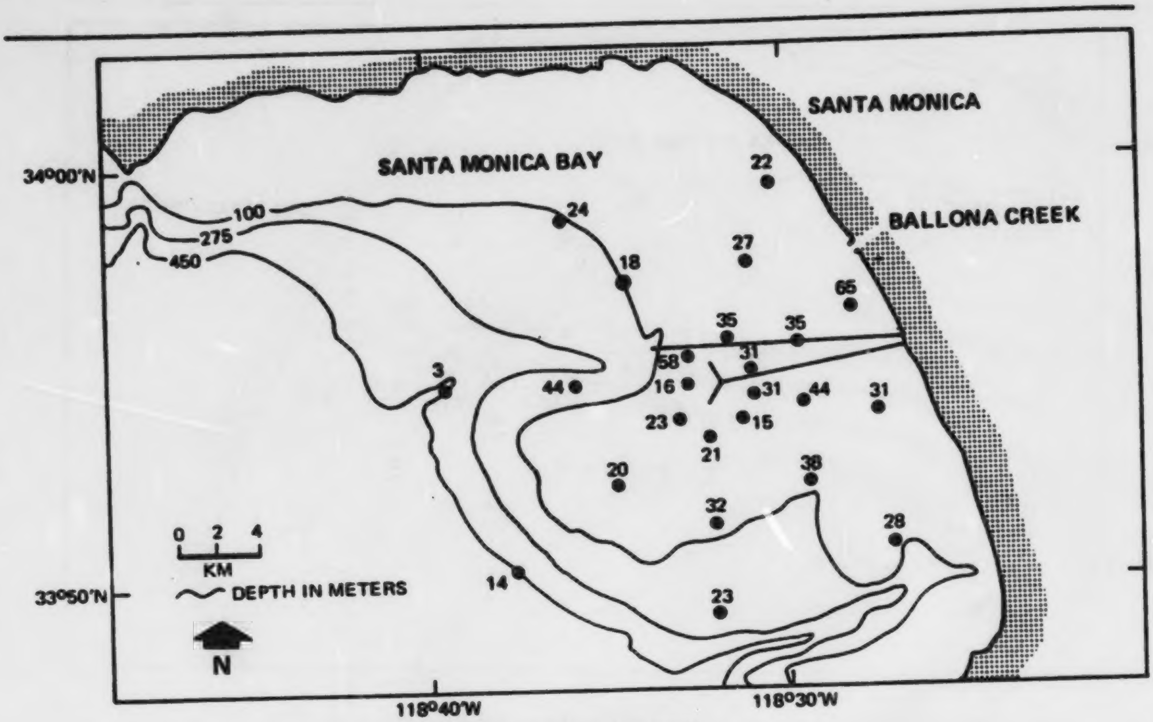
BALLONA CREEK
B13
B14
B12
B5
B4
B3
B9
B10
B8
B11
B7
B6
B15
SMC
B16
B2
B17
0 2 4
KM
33°50'N
DEPTH IN METERS
N
118°40'W
118°30'W

Figure 4 Box Core Collection Locations Around the Hyperion Outfall System, July 1971.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SANTA MONICA BAY
BALLONA CREEK
34°00'N
33°50'N
118°40'W
118°30'W
100
275
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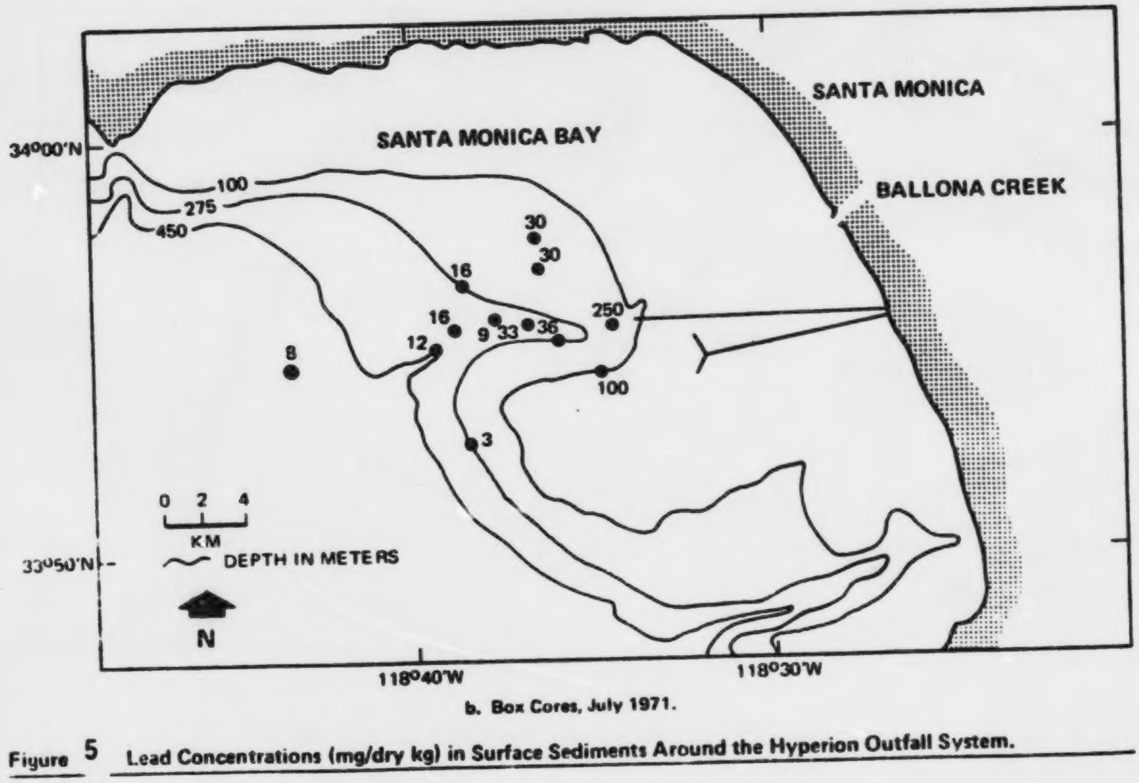
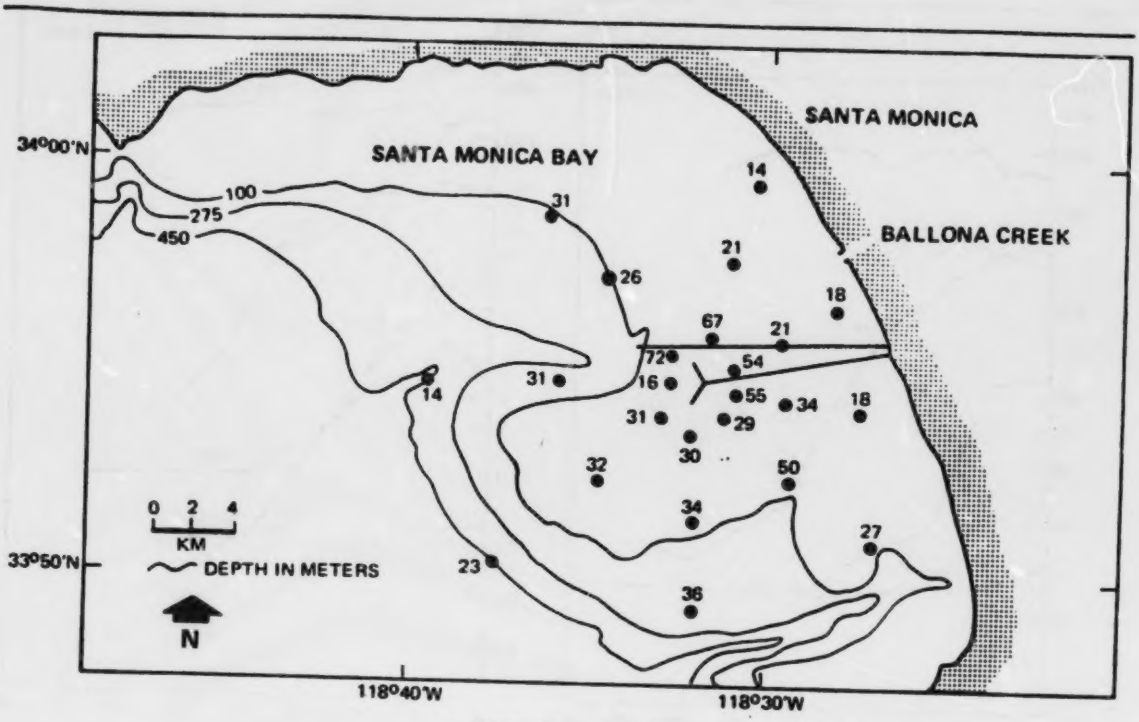
22
24
27
18
65
35
35
58
31
3
44
16
31
44
31
23
15
21
20
38
32
28
14
23
0 2 4
KM
DEPTH IN METERS
N

a. Phleger Cores, June 1970.


SANTA MONICA
SANTA MONICA BAY
BALLONA CREEK
34°00'N
33°50'N
118°40'W
118°30'W
100
275
450
30
30
16
16
9
33
36
250
8
12
100
3
0 2 4
KM
DEPTH IN METERS
N

b. Box Cores, July 1971.

Figure 5 Lead Concentrations (mg/dry kg) in Surface Sediments Around the Hyperion Outfall System.


SANTA MONICA BAY
SANTA MONICA
BALLONA CREEK
34°00'N
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DEPTH IN METERS
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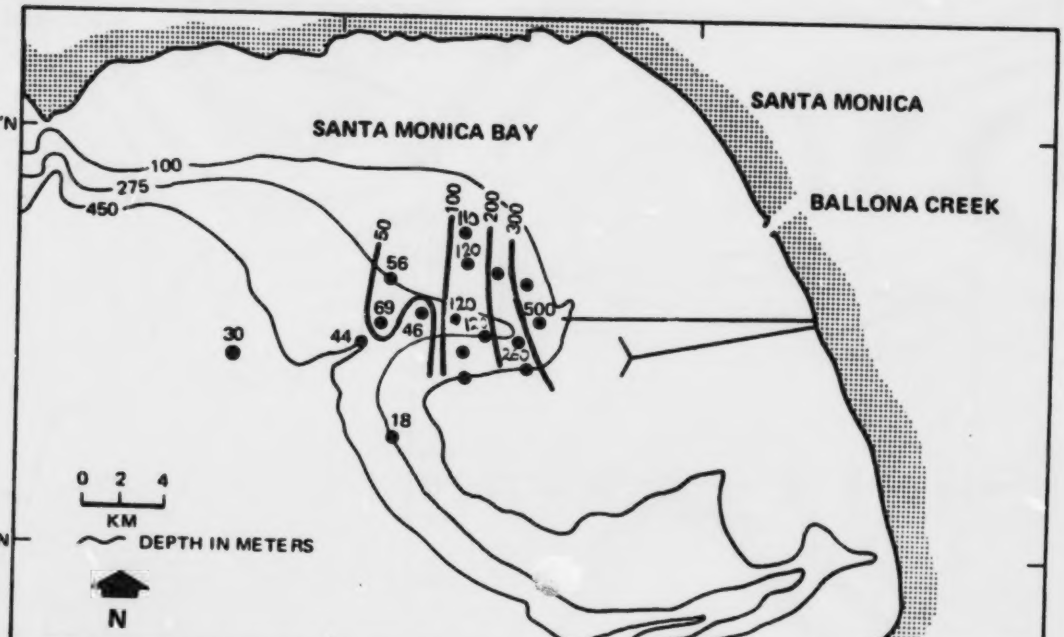
a. Phleger Cores, June 1970.


SANTA MONICA BAY
SANTA MONICA
BALLONA CREEK
34°00'N
33°50'N
118°40'W
118°30'W
100
275
450
50
100
200
300
115
120
56
69
46
44
30
120
123
500
260
18
0 2 4
KM
DEPTH IN METERS
N

b. Box Cores, July 1971.

Figure 6 Copper Concentrations (mg/dry kg) in Surface Sediments Around the Hyperion Outfall System.

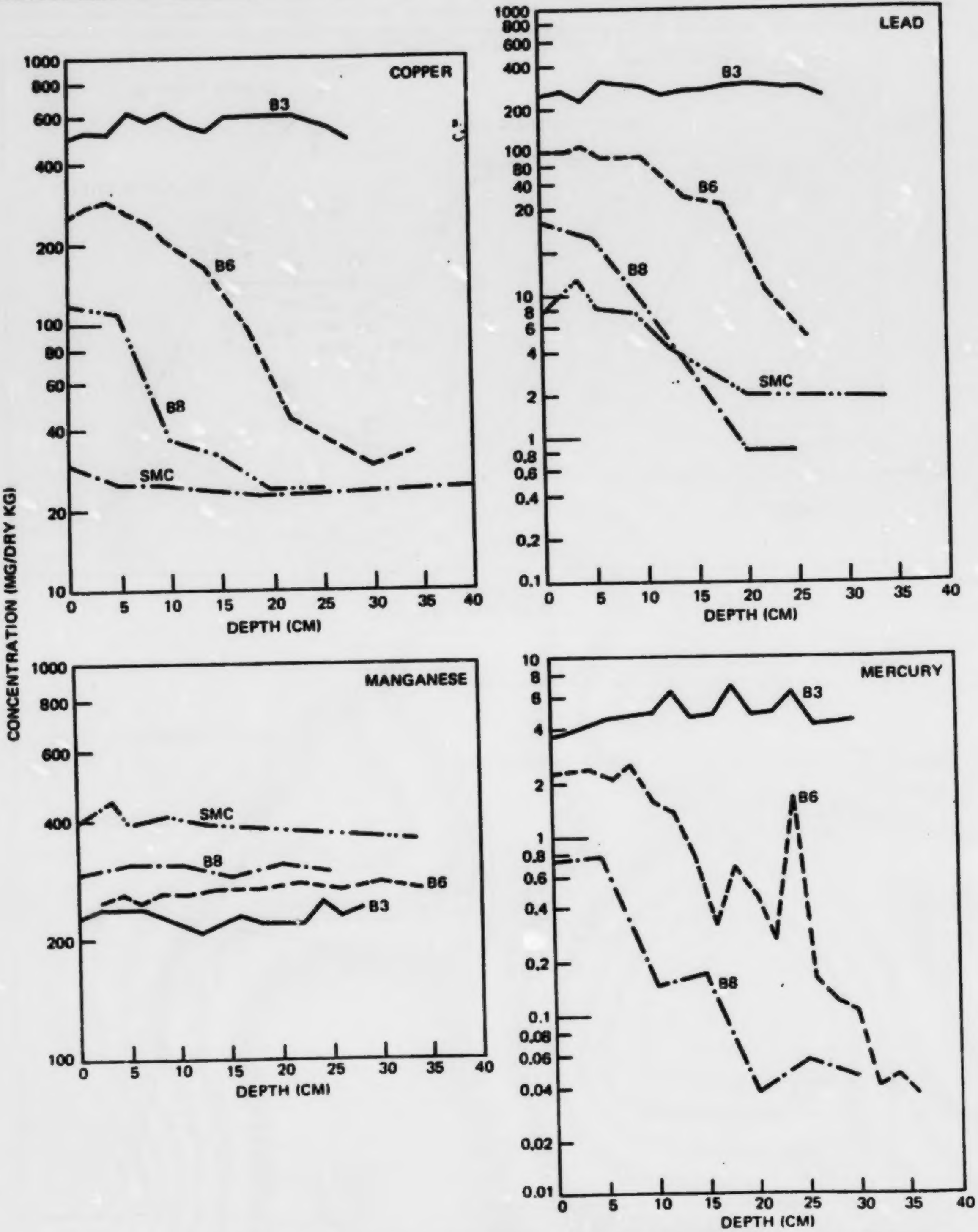


Figure 7 Vertical Profiles of Metal Concentrations in Sediments Collected in Santa Monica Canyon off the Hyperion Sludge Outfall (Box Cores, July 1971). Station Locations Shown in Figure

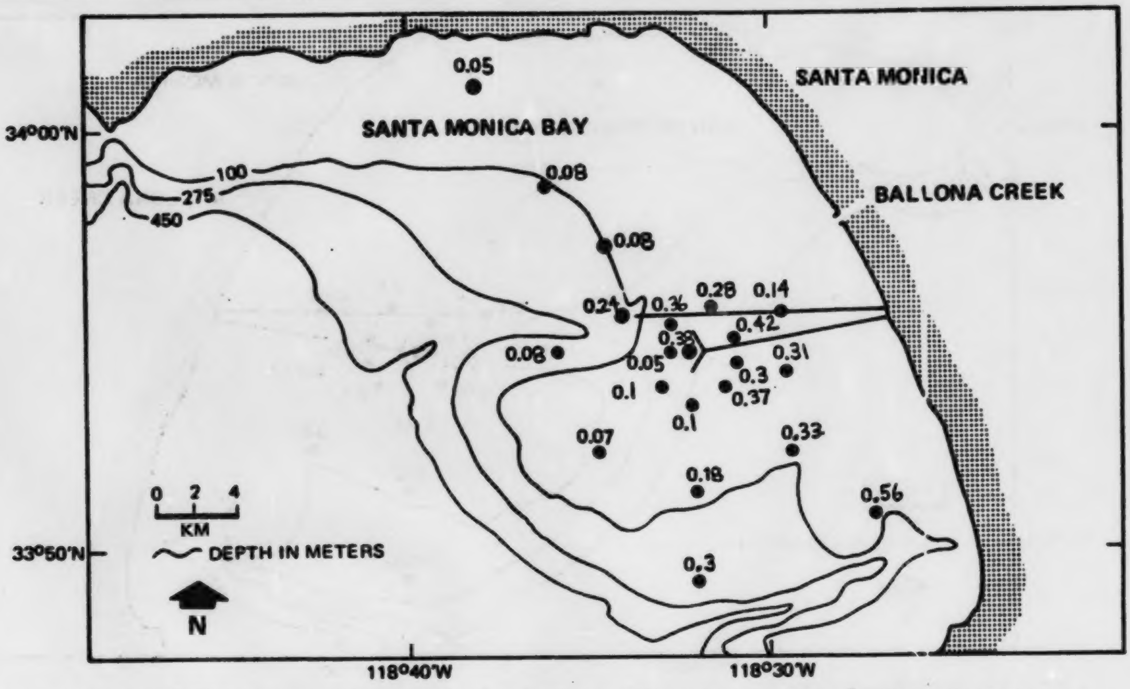

SANTA MONICA
SANTA MONICA BAY
BALLONA CREEK
34°00'N
33°50'N
118°40'W
118°30'W
100
275
450
0.05
0.08
0.08
0.24
0.36
0.28
0.14
0.42
0.32
0.08
0.05
0.31
0.3
0.1
0.37
0.1
0.07
0.33
0.18
0.56
0.3
0 2 4
KM
DEPTH IN METERS
N

Figure 8 Total DDT Concentrations (mg/dry kg) in Surface Sediments Around the Hyperion Outfall System (Phleger Cores, July 1972).

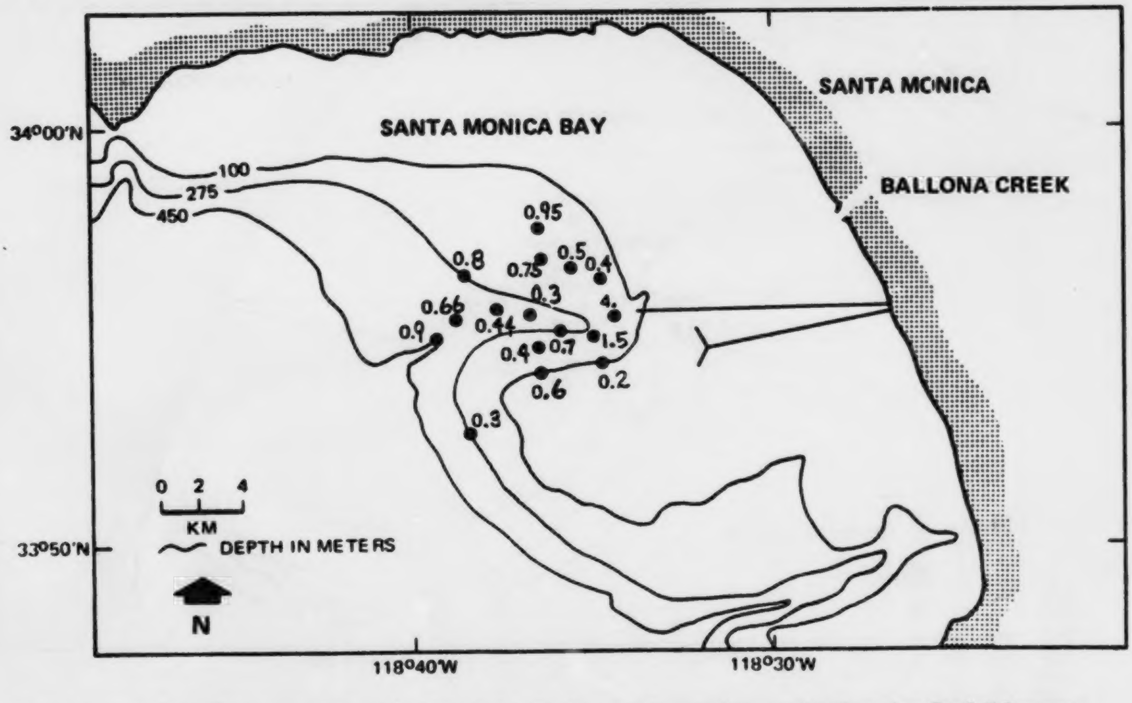

SANTA MONICA
SANTA MONICA BAY
BALLONA CREEK
34°00'N
33°50'N
118°40'W
118°30'W
100
275
450
0.95
0.8
0.75
0.5
0.4
0.3
0.66
0.44
4.
0.9
1.5
0.4
0.7
0.6
0.2
0.3
0 2 4
KM
DEPTH IN METERS
N

Figure 9 Total DDT Concentrations (mg/dry kg) in Surface Sediments Around the Hyperion Outfall System (Box Cores, July 1971).

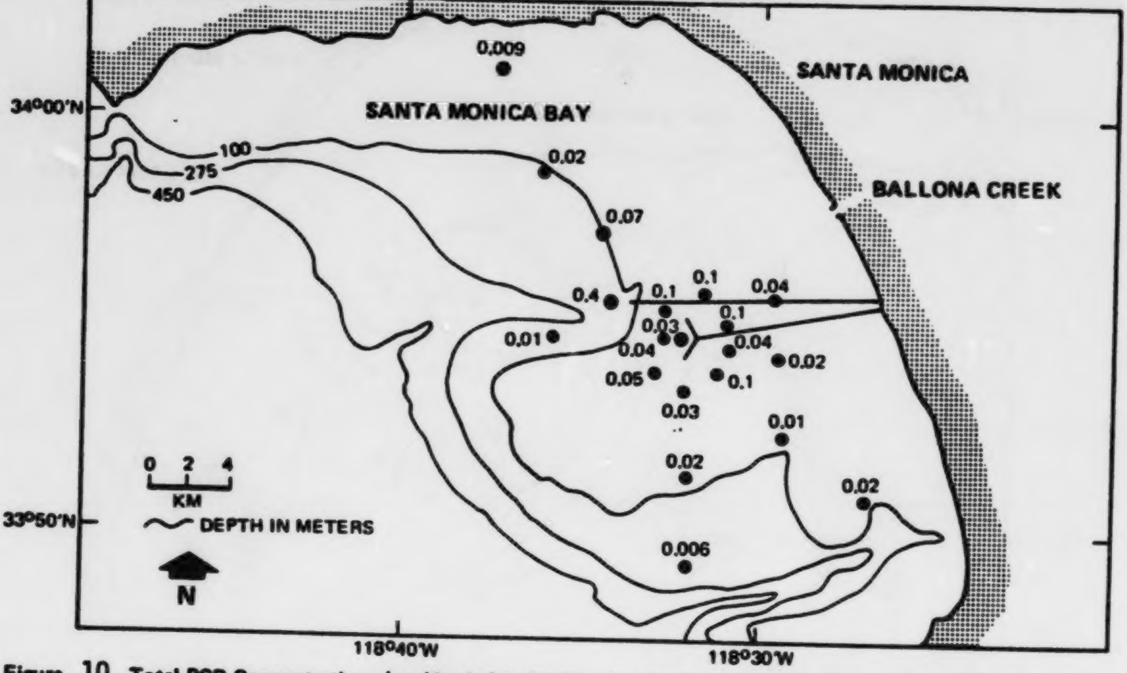

SANTA MONICA
SANTA MONICA BAY
BALLONA CREEK
34°00'N
33°50'N
118°40'W
118°30'W
100
275
450
0.009
0.02
0.07
0.4
0.1
0.1
0.04
0.1
0.03
0.01
0.04
0.04
0.02
0.05
0.1
0.03
0.01
0.02
0.02
0.006
0 2 4
KM
DEPTH IN METERS
N

Figure 10 Total PCB Concentrations (mg/dry kg) in Surface Sediments Around the Hyperion Outfall System (Phleger Cores, July 1972).

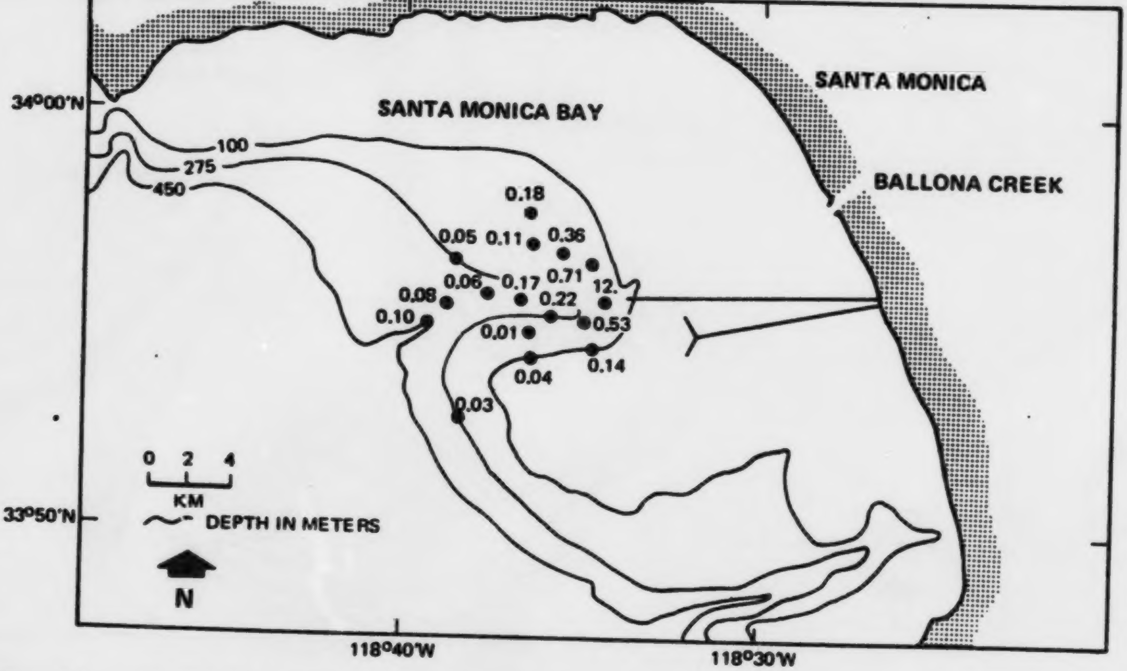

SANTA MONICA
SANTA MONICA BAY
BALLONA CREEK
34°00'N
33°50'N
118°40'W
118°30'W
100
275
450
0.18
0.05
0.11
0.36
0.71
0.06
0.17
12.
0.08
0.22
0.10
0.01
0.53
0.04
0.14
0.03
0 2 4
KM
DEPTH IN METERS
N

Figure 11 Total PCB Concentrations (mg/dry kg) in Surface Sediments Around Hyperion Outfall System (Box Cores, July 1971).

oxygen, or highly organic substrate. The results of a 1972 survey of the polychaetes is Santa Monica Bay sediment samples are presented in Figure 12, in which the relative abundance of capitellids is recorded for each station sampled. High proportions (16% - 58%) of capitellids were found in the sediments in the vicinity of the two outfalls, and lower percentages (1% - 7%) were observed in areas further from the outfalls. An indication of the biological enrichment potential of the sludge discharge was also revealed by this same study. The station at the end of the sludge outfall had a polychaete biomass of 69 g/sq m while the station between the two Y diffusers had a polychaete biomass of 11 g/sq m.

Although the actual causes of fin erosion diseases in benthic fish has not yet been determined, there are indications that the disease is related in some way to the discharge of wastewater solids. Fin erosion in Dover sole has been observed in catches from 5 out of 25 locations in Santa Monica Bay sampled in 1971. In each case the incidence of disease was greater than ten percent, and 4 of 5 locations were within 2 miles (3.2 km) of the sludge discharge. The other location was 7 miles (11.3 km) northwest of the discharge.

In addition to sampling and analyses techniques, visual observation of the ocean bottom and the marine life there is a valuable technique for assessing the environmental consequences of ocean discharges. Recent visual inspection of the ocean bottom using undersea television equipment in the vicinity of the sludge outfall has provided the following qualitative description of the area. The pipe itself provides a solid substrate upon which sea anemones grow in extremely high densities. The most abundant large organisms in the vicinity of the outfall are starfish and their population density is highest in the sediments adjacent to the pipeline. Sea urchins and Dover sole are commonly seen while other fishes and animals such as crabs are seen less frequently.

Summary and Conclusions

Quantitative analyses of the two Hyperion discharges to Santa Monica Bay show that the mass emission rates of the sludge discharge are similar to those of the combined primary and secondary effluents. Physical and chemical properties of the ocean sediments in the bay have been altered by the waste discharges, with the greatest effects observed in Santa Monica Canyon and closest to the sludge outfall. Depth profiles of sediment heavy metals concentrations indicate that the depths of significant quantities of sludge particulates are greater than 1 foot (30.5 cm) near the sludge outfall and 1 foot or less at distances greater than 2 miles (3.2 km) down canyon from the sludge discharge. Biological evidence of environmental stress in the vicinity

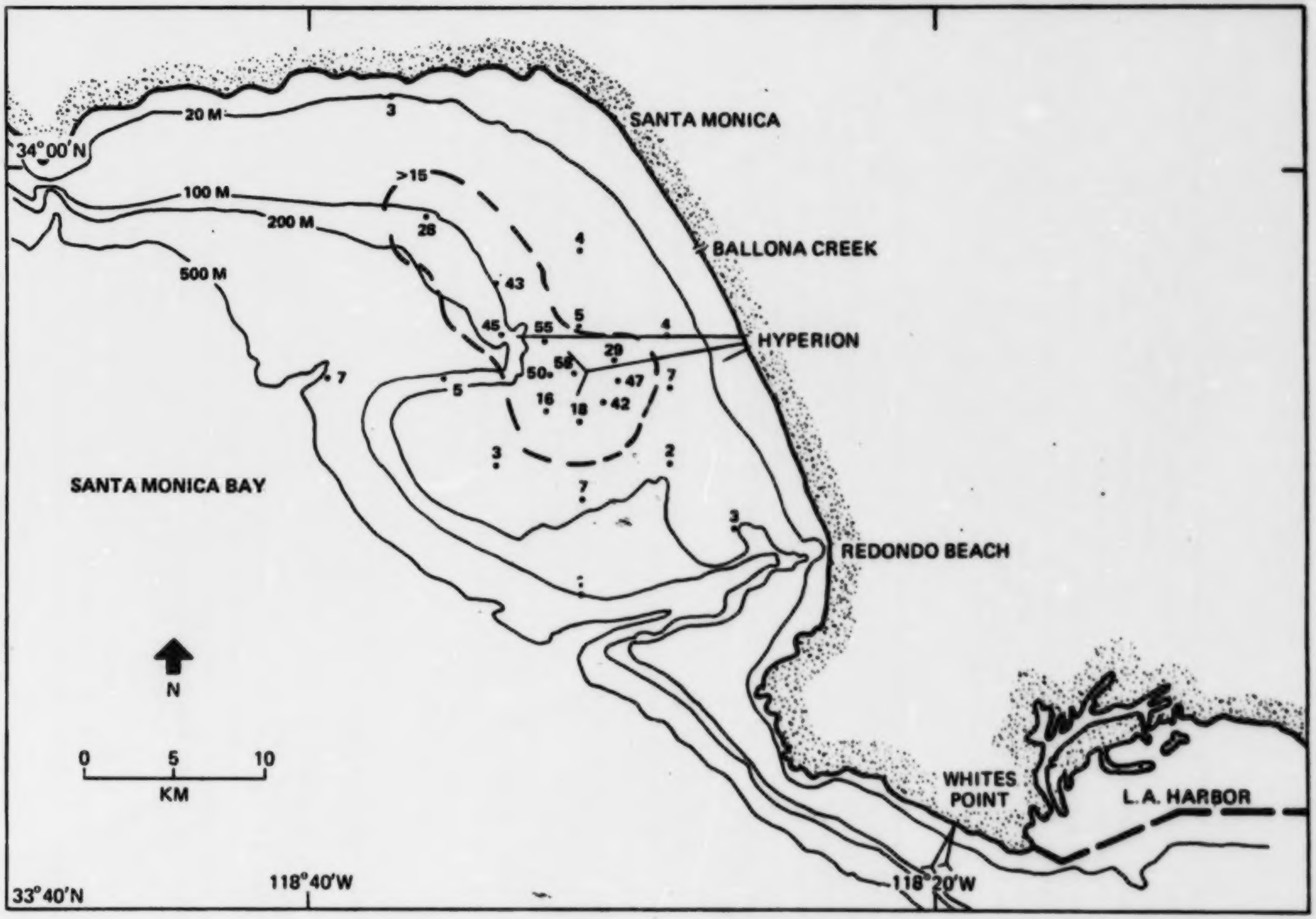

SANTA MONICA
BALLONA CREEK
HYPERION
REDONDO BEACH
WHITES POINT
L.A. HARBOR
SANTA MONICA BAY
34°00'N
33°40'N
118°40'W
118°20'W
20 M
100 M
200 M
500 M
>15
3
28
4
43
5
45
55
4
29
50
56
47
7
16
18
42
7
5
3
2
7
3
N
0
5
10
KM

Fig. 12 Relative Abundance of Capitellids (Percent of Total Polychaete Specimens) in Santa Monica Bay, January - February 1972.

of the discharges has been found, but the areas affected are by no means biological deserts.

Research concerned with the environmental effects of wastewater and sludge discharges to the open ocean is continuing on the basis that only by understanding the consequences fo each alternative disposal practice can rational decisions be made. The California State Water Quality Control Board has taken the following position with respect to sludge discharge to the ocean:

> "The discharge of municipal and industrial waste sludge and sludge digester supernatant directly to the ocean, or into a waste stream that discharges to the ocean without further treatment, shall be prohibited." (2)

The difficulty with such a position is that the sludge continues to be produced and must be disposed of in one way or another. Due to the nature of the material, any disposal practice is likely to produce environmental stress in the area of disposal. If the solution to the sludge disposal problem is constrained to be one which minimizes adverse environmental effects, it would be best to have as many viable alternatives open for evaluation as possible.

Acknowledgements

Much of the data presented in this paper has been obtained from the Hyperion Laboratory and certainly almost all of the information has been aided by the cooperation of Hyperion personnel.

Figure 3-12 are exerpted from reference (1).

References

1. "The Ecology of the SOuthern California Bight: Implications for Water Quality Management", Southern California Coastal Water Research Project (SCCWRP) Three Year Report, March 1973.

2. "Water Quality Control Plan for Ocean Waters of California", California State Water Resources Control Board, Sacramento, California. Adopted and effective July 6, 1972.

A DISCUSSION OF
A NEAR FIELD DISPERSION MODEL
FOR SLUDGE DISCHARGES TO COASTAL WATERS

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Coastal Pollution Branch
Pacific Northwest Environmental Research Laboratory
National Environmental Research Center
Environmental Protection Agency
Corvallis, Oregon 97330

April 1974

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In preparing this document it became apparent that use of a consistent system of units would present a problem. The computer program utilizes both metric and English units. Thus, to maintain continuity the units required as input and displayed in the output are used here.

INTRODUCTION

Barging of waste materials to offshore dumpsites is a disposal method which is being used in this country for a variety of wastes that include industrial liquids and sludges, sewage sludge, and dredge spoils. This practice is controlled, in part, by a permit system, which for industrial wastes and sewage sludge, is administered by the Environmental Protection Agency.

Investigators, in examining the effects of past ocean dumping, have demonstrated ecological alterations from such practices. Periodic monitoring is, or should be, required for all wastes discharged to coastal waters to enable undesired changes or effects to be recognized, and corrective action taken to preclude irreversible ecological damage.

The Coastal Pollution Branch of EPA's Pacific Northwest Environmental Research Laboratory sponsored the development of a generalized mathematical model to aid in the prediction of the physical fate of waste materials barged to coastal waters. Prediction models of this type may be useful for both monitoring and permit application evaluations. Computer simulation techniques provide a detailed description of the sequence of events that occur after discharge, enhancing the interpretation of physical mechanisms.

The objective of this paper is a demonstration of the application of this simulation technique for the problem of sewage sludge discharges. For given inputs, the results are compared for representative environmental conditions. The depth of penetration of the waste cloud, the dilution, and the depositional characteristics of settleable and floating solids are also discussed and related to pertinent federal regulations governing ocean dumping.

The mathematical model, developed by Dr. R. C. Y. Koh and Y. C. Chang under an EPA research grant to the Tetra Tech Inc., Pasadena, CA is contained in an EPA report (1) published in December of 1973 and will be described here only conceptually.

The model is descriptive of three separate methods of discharge representing instantaneous, pumped, and wake releases. The waste may be characterized by a liquid and a maximum of eight distinct solid or particulate phases. Each phase is traced through three computer linked transport modes as represented in Figure 1. These transport modes are also shown on the program flow diagram, Figure 2.

The first transport mode is a convective descent, terminated when the material either encounters the bottom or reaches a position of buoyant equilibrium within the water column (i.e. vertical velocity equals zero). In the latter case a dynamic vertical collapse occurs when a density difference exists between the bulk waste cloud and the ambient fluid. The final transport mode is one of passive long term diffusion where the vertical concentration distribution, areal size, and amount of accumulated solids are determined.

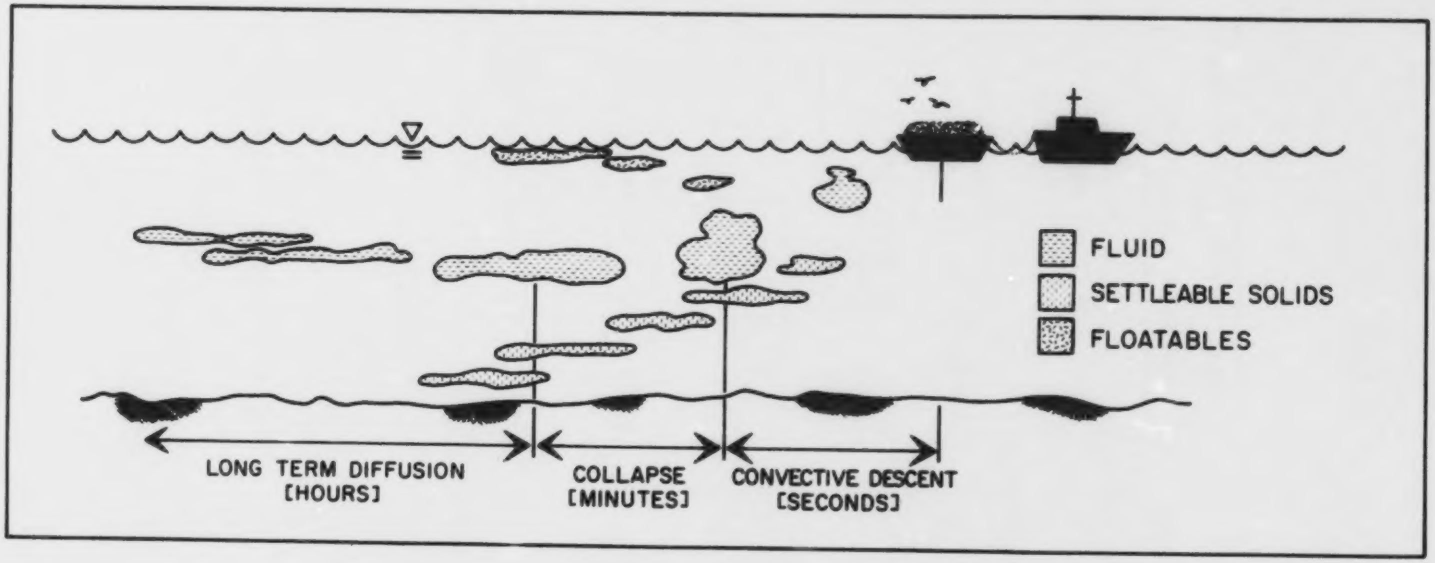
FLUID
SETTLEABLE SOLIDS
FLOATABLES
LONG TERM DIFFUSION [HOURS]
COLLAPSE [MINUTES]
CONVECTIVE DESCENT [SECONDS]

Fig. 1 Basic Transport Modes; Barge Operation 1

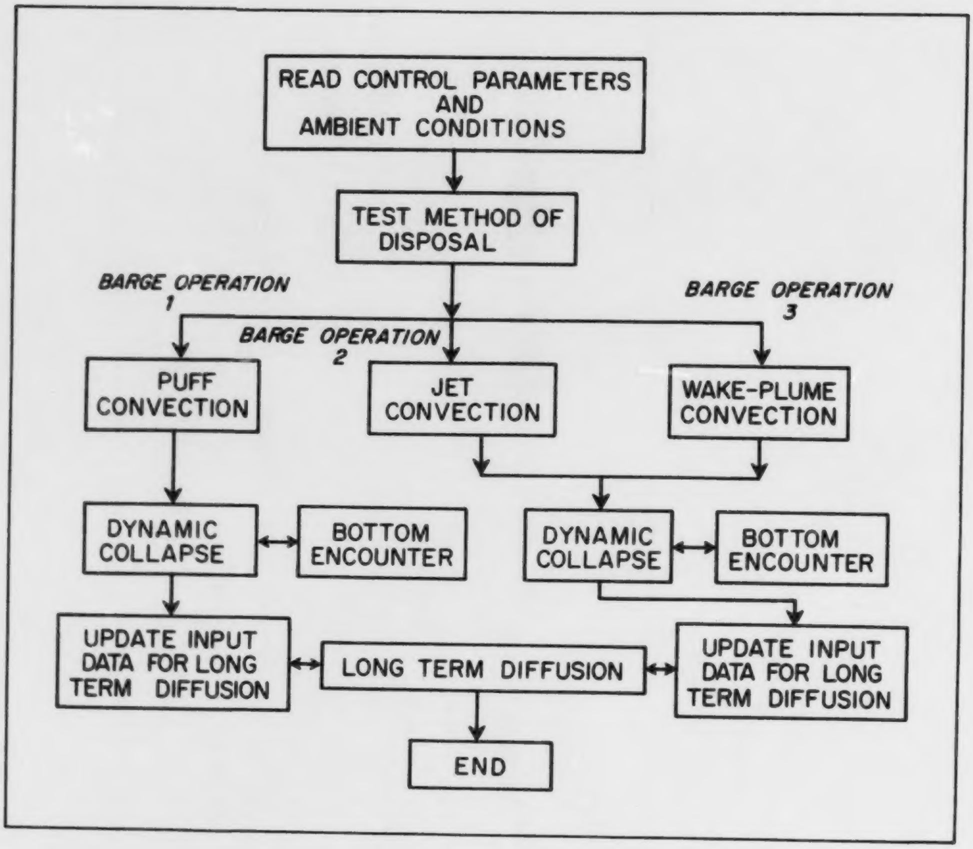
READ CONTROL PARAMETERS AND AMBIENT CONDITIONS
TEST METHOD OF DISPOSAL
BARGE OPERATION 1
BARGE OPERATION 2
BARGE OPERATION 3
PUFF CONVECTION
JET CONVECTION
WAKE-PLUME CONVECTION
DYNAMIC COLLAPSE
BOTTOM ENCOUNTER
DYNAMIC COLLAPSE
BOTTOM ENCOUNTER
UPDATE INPUT DATA FOR LONG TERM DIFFUSION
LONG TERM DIFFUSION
UPDATE INPUT DATA FOR LONG TERM DIFFUSION
END

Fig. 2 Computer Model Flow Diagram

PROBLEM DEFINITION

The basic problem examined in the subsequent examples relate directly to the periodic discharge of sewage sludge to typical coastal waters. Tables 1 and 2 provide the model input requirements necessary to characterize the barge and waste. Figure 3 depicts the environmental conditions for the site. The parameters were chosen from available field data and represent seasonal differences likely to be encountered in North temperature coastal climates.

BARGE CHARACTERISTICS

The barge characteristics are representative of typical sludge barging operations. Many operations, however, may have discharge times or waste volumes that deviate significantly from these cited values.

The model does not simulate the interactions of multi-port discharges, and the situation described in Table 1, where twelve ports are shown will be treated as a single port where only 1/12 of the total barge load is examined. These results must be proportionately scaled when a need to account for the entire waste loading exists.

WASTE CHARACTERISTICS

The waste is a digested municipal sewage sludge characterized by three separate phases; a liquid, a settleable solid, and a floating solid. The specific characteristics required by the model are presented in Table 2.

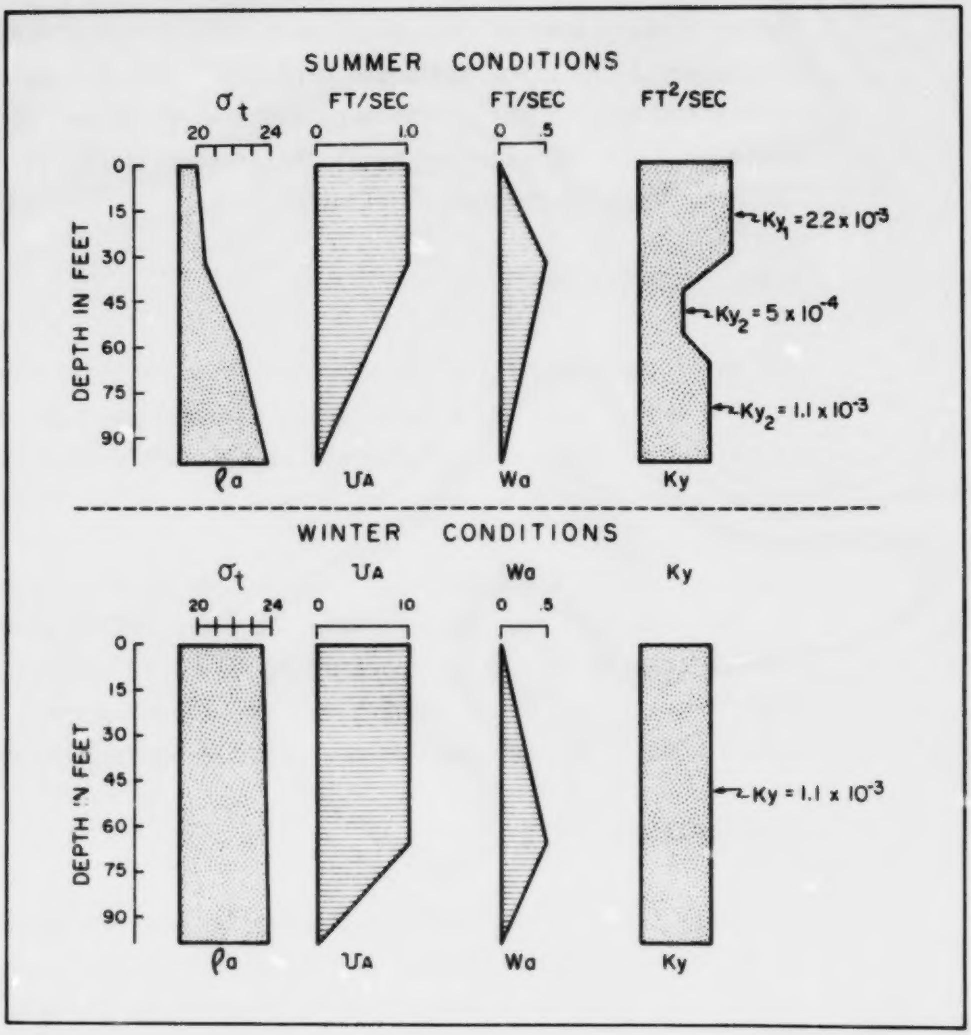

Fig. 3 Site Environmental Conditions

SITE CHARACTERISTICS

The environmental conditions at a proposed disposal site must be characterized in terms of vertical profiles of density, current, and coefficients of vertical eddy diffusivity. The coefficients used in these examples are based on a relationship developed by Koh and Fan (3) where Ky is inversely proportional to the ambient density gradient as given by the following equation:

$$Ky = B/E \qquad 1.1$$

where

Ky = coefficient of vertical eddy diffusivity [ft^2/sec]
E = density gradient expressed as [gms/ft-cc]
B = constant = 1.1×10^{-7} [ft-gms/cc-sec]

Table 1

Barge Discharge Characteristics

Input Term	Model Req. Units	Metric Units
Barge Width	60 ft	18.29 m
Barge Speed	5.0 ft/sec	152.40 cm/sec
Discharge time	600.0 sec	600.0 sec
Direction of travel*	180°	180°
Barge Length	200 ft	60.96 m
Angle of Discharge**	0°	0°
Number of Ports	12.0	12.0
Depth of Ports	10.0 ft	304.8 cm
Diameter of Ports	1.16 ft	35.36 cm
Velocity of Discharge	4.227 ft/sec	128.84 cm/sec

*relative to + X axis

**relative to vertical axis where 0° denotes a vertical downward discharge

Table 2

Waste Characteristics

Input Term	Model Req. Unit	Metric Units
Bulk Specific Gravity	1.040 gms/cc	1.040 gms/cc
Number of Solid Phases	2	2
Density of settleable solids	1.61 gms/cc	1.61 gms/cc
Conc. (by volume)	4.5%	4.5%
Settling Velocity	0.005 ft/sec	0.1524 cm/sec
Density of floating solids	0.98 gms/cc	0.98 gms/cc
Conc. (by volume)	0.5%	0.5%
Settling velocity	0.05 ft/sec	1.524 cm/sec

DISCUSSION OF MODEL RESULTS

PENETRATION AND DILUTION

The output from the program describing the convective descent and dilution of the waste materials is usually descriptive of effects for the un-separated waste components but in some cases reveals constituent separations caused by settling solids and floatable particles.

The results for the first two transport modes are presently graphically in Figures 4, 5, and 6. Figure 4 shows the depth of penetration and the size of the resulting plume for a single port release of identical wastes under seasonal differences in the current and density gradients. The summer conditions result in a thermocline

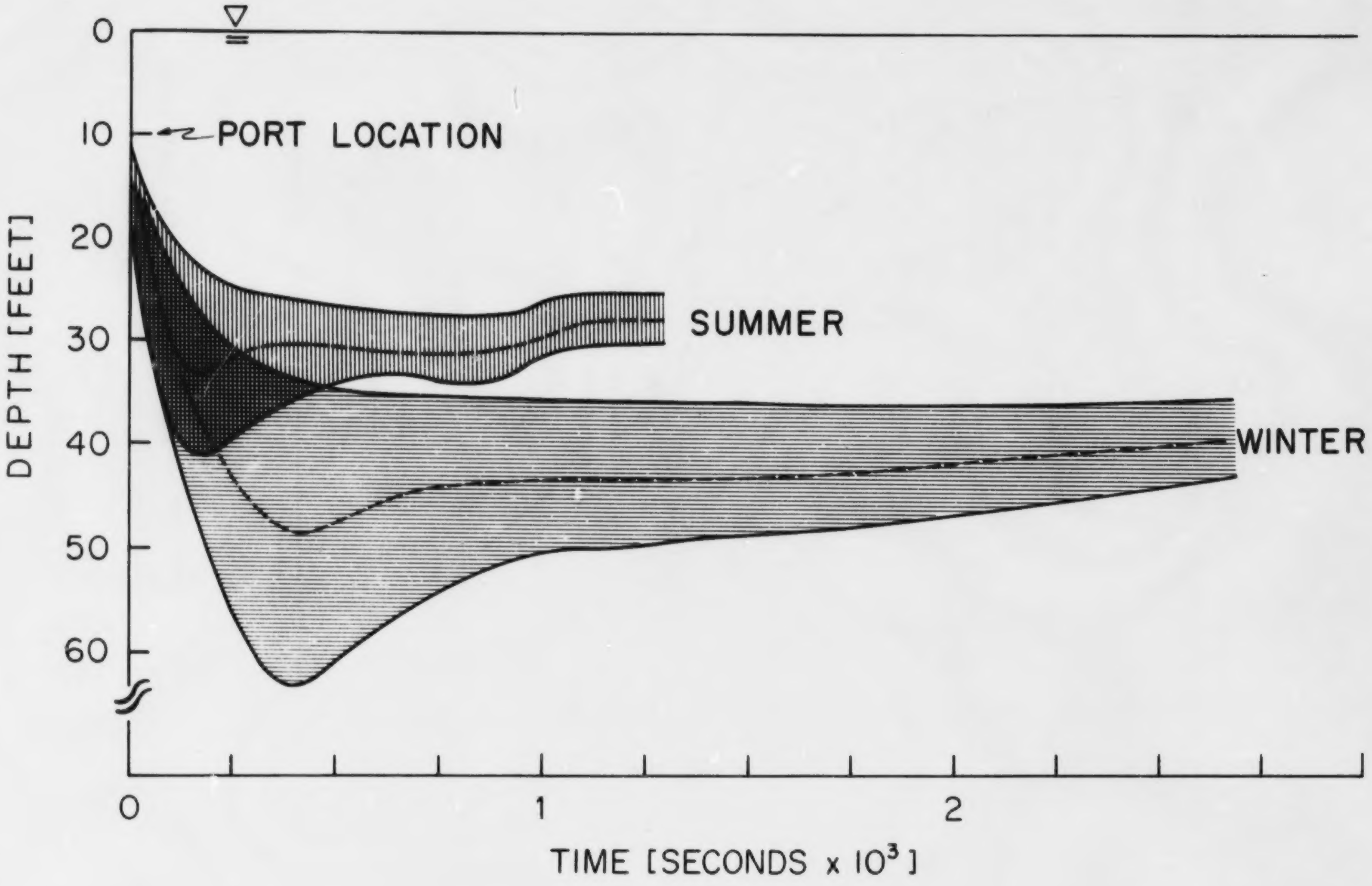

Fig. 4 Waste Cloud Penetration During Convective Descent and Collapse Transport Modes

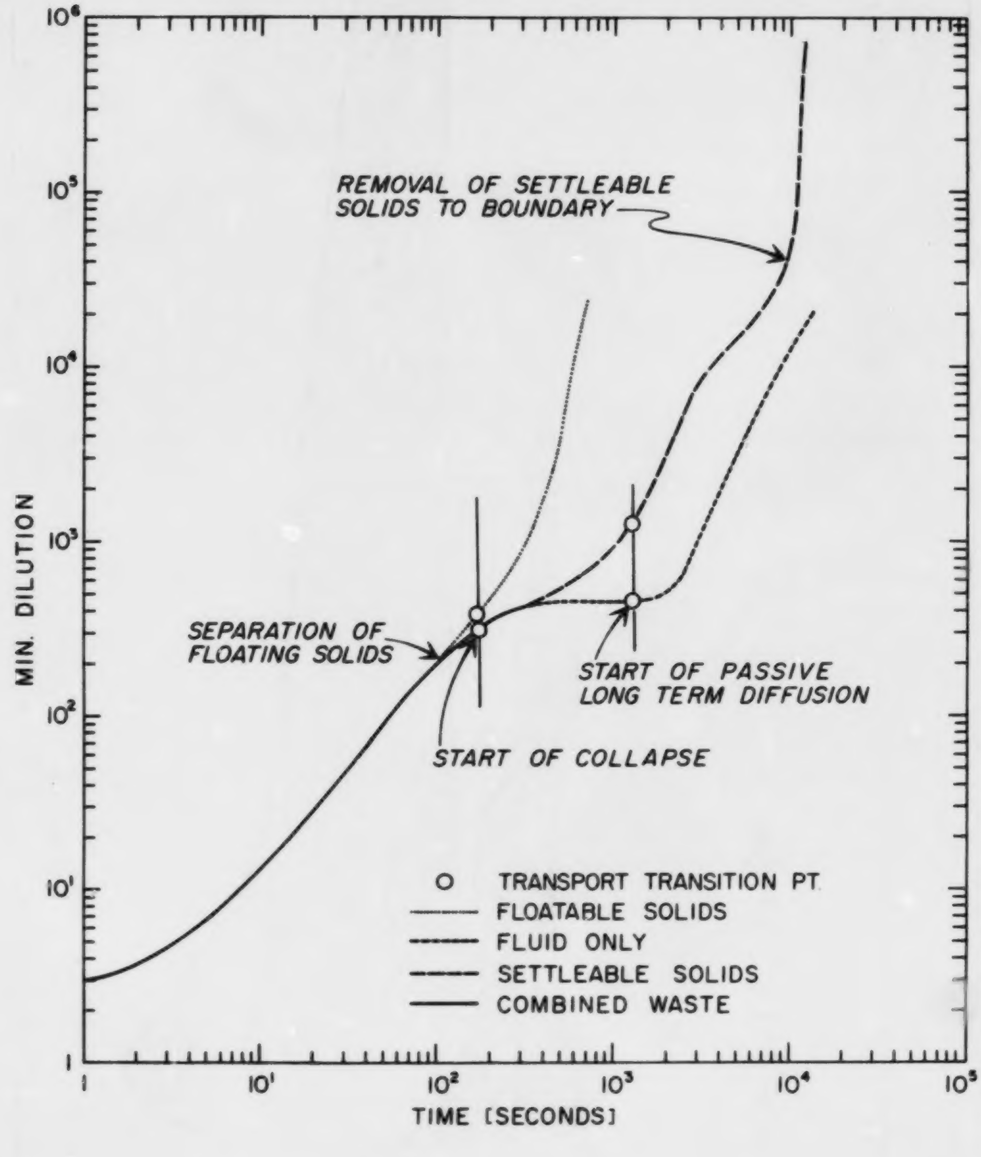

Fig. 5 Minimum Single Port Dilution for Each Waste Phase as Functions of Time; Summer Conditions

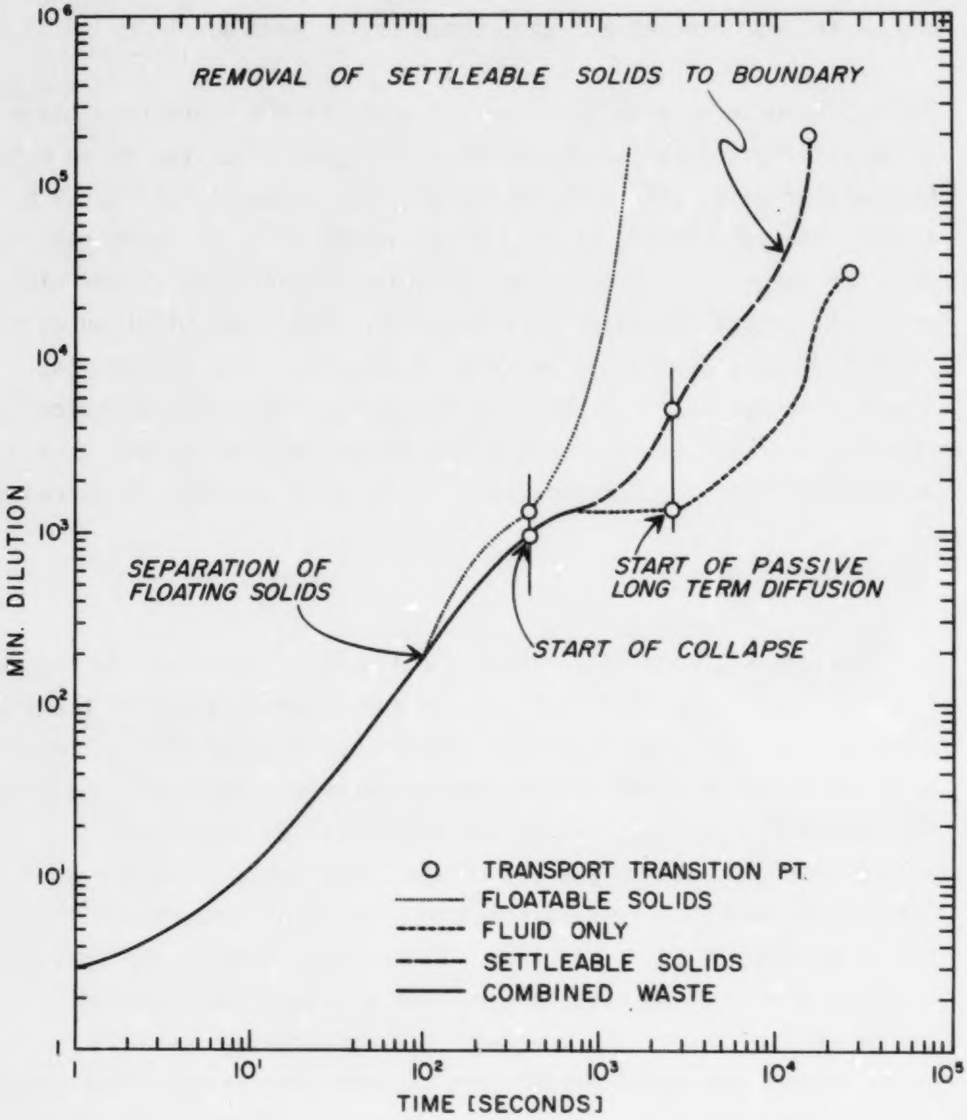

Fig. 6 Minimum Single Port Dilution for Each Waste Phase as Functions of Time; Winter Conditions

which restricts the penetration of the waste cloud. The winter conditions, with only a slight density gradient, result in buoyant equilibrium deep within the water column. If no gradient existed the wastes would penetrate unhindered to the bottom.

During the descent mode the flux of solids in the descending cloud is decreasing. This is not apparent in Figure 4 but can be seen in the tabular data, and in the dilution rates as shown in Figures 5 and 6. The separation, or branching, of the dilution curve results from the removal of solids from the cloud as particles either sink or float, or are deposited on a boundary. The rapid dilution of the floatable phase should not be cause to dismiss these pollutants. A more detailed examination would reveal that the trend is quickly reversed when the solids collect and concentrate on the boundary and become available for further transport by wind and ambient currents.

LONG TERM DIFFUSION

The physical sequence after dynamic collapse is presented in Figures 7 and 8. Here, the vertical position and lateral extent of the waste cloud are presented at four time steps. The figures depict the horizontal planes where maximum waste concentrations occur and show the separation of floating and settleable solids and their ascent-descent paths to the boundaries. The lateral dimensions calculated from the computed standard deviations account for 95 percent of the waste load. These plots showing the spatial location of the three waste phases over a several hour period can be used to verify the adequacy of a monitoring operation or to compare the model predictions against regulations that prescribe maximum permissible concentrations and mixing volumes. The summer case,

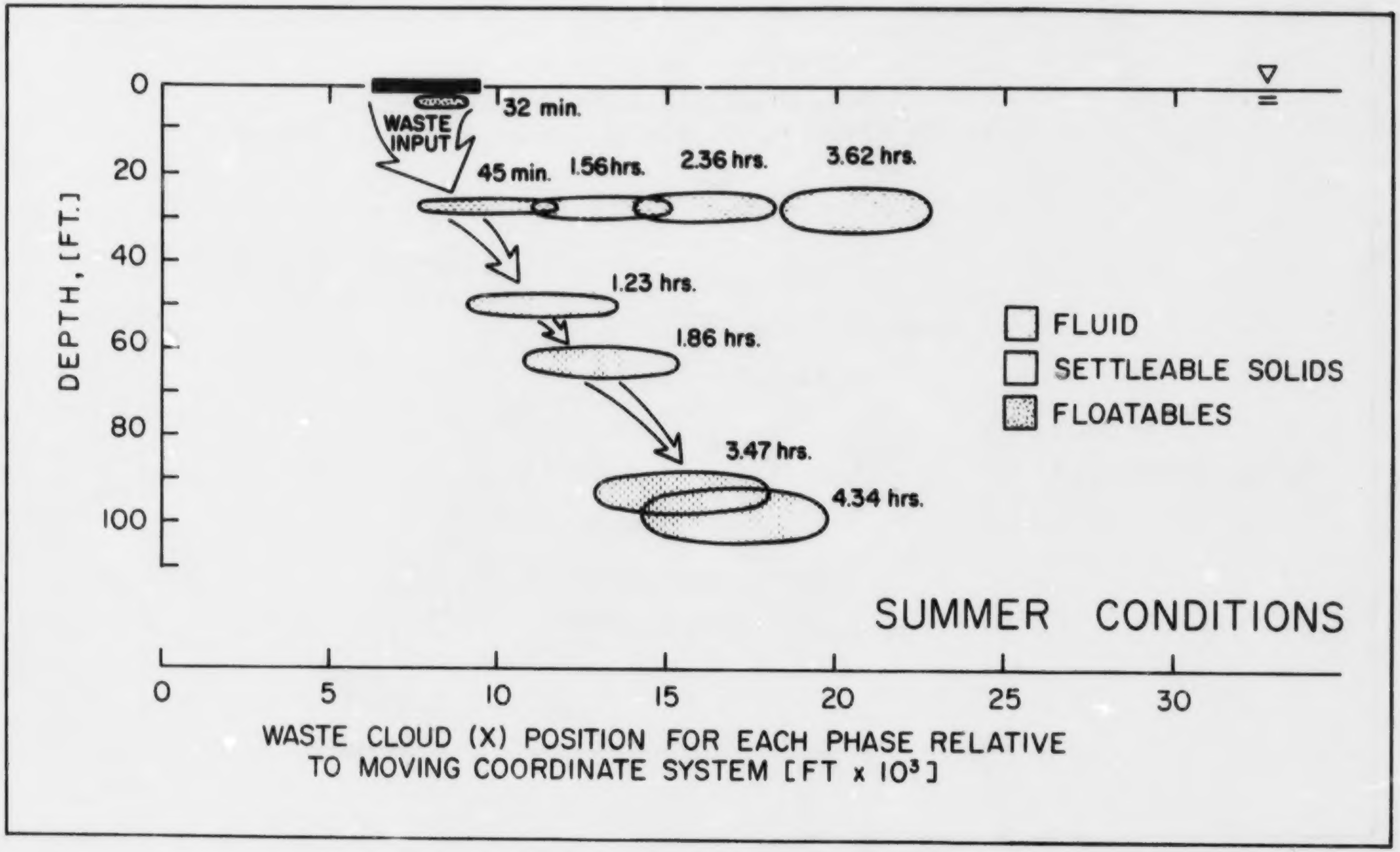

Fig. 7 Time Series View of Long Term Diffusional Transport; Summer Conditions

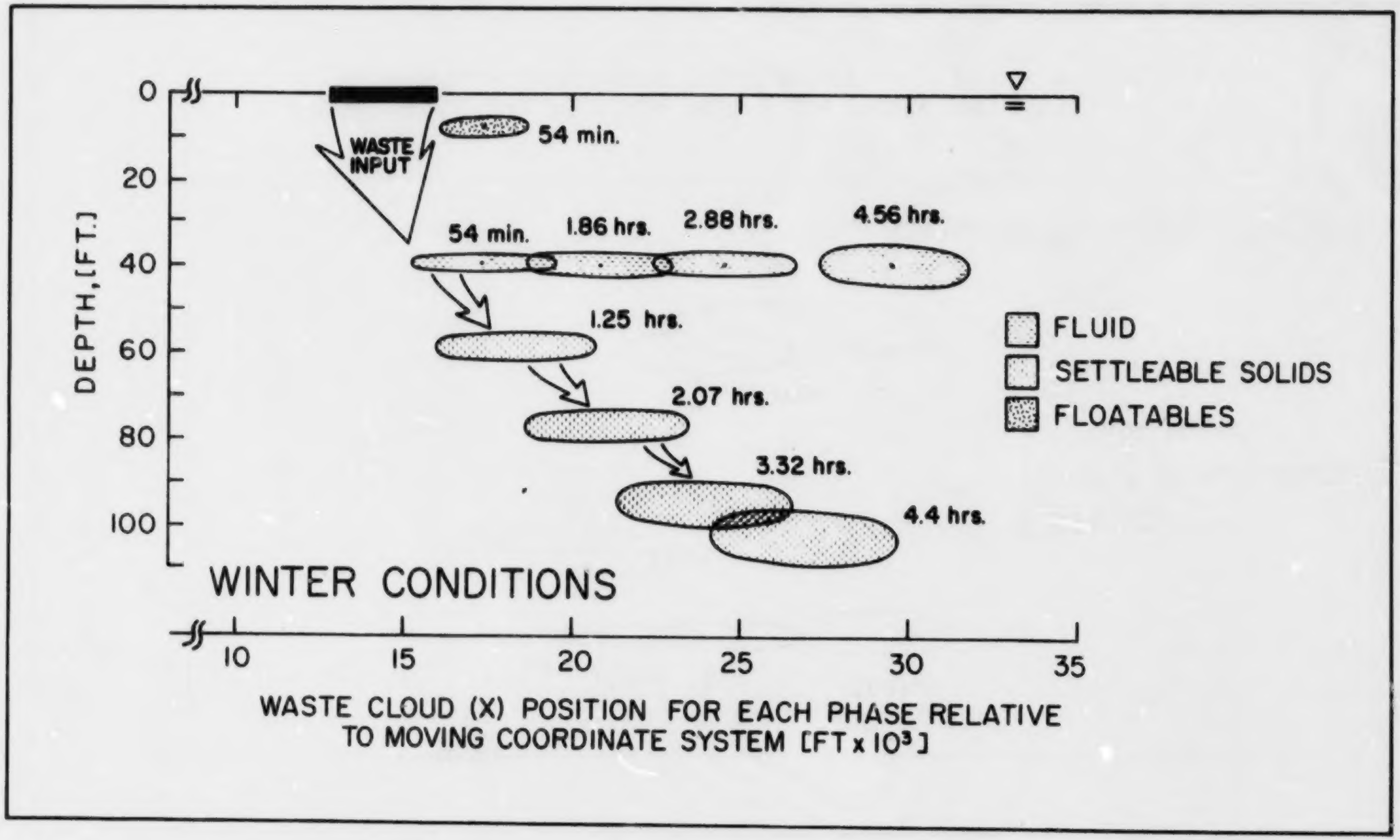

Fig. 8 Time Series View of Long Term Diffusional Transport; Winter Conditions

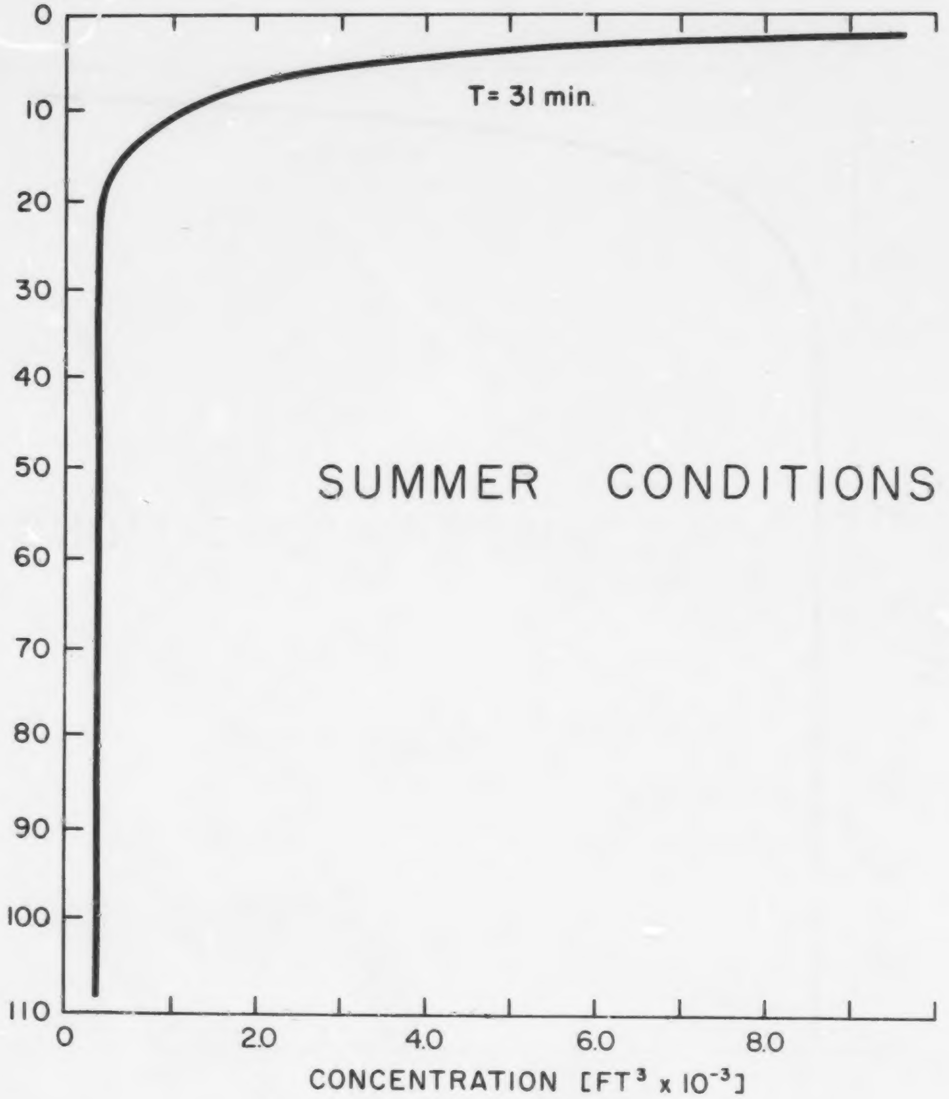

Fig. 9 Vertical Distribution of Relative Wastes Concentrations During Long Term Diffusional Transport; Floatable Solid Phase; Summer Conditions

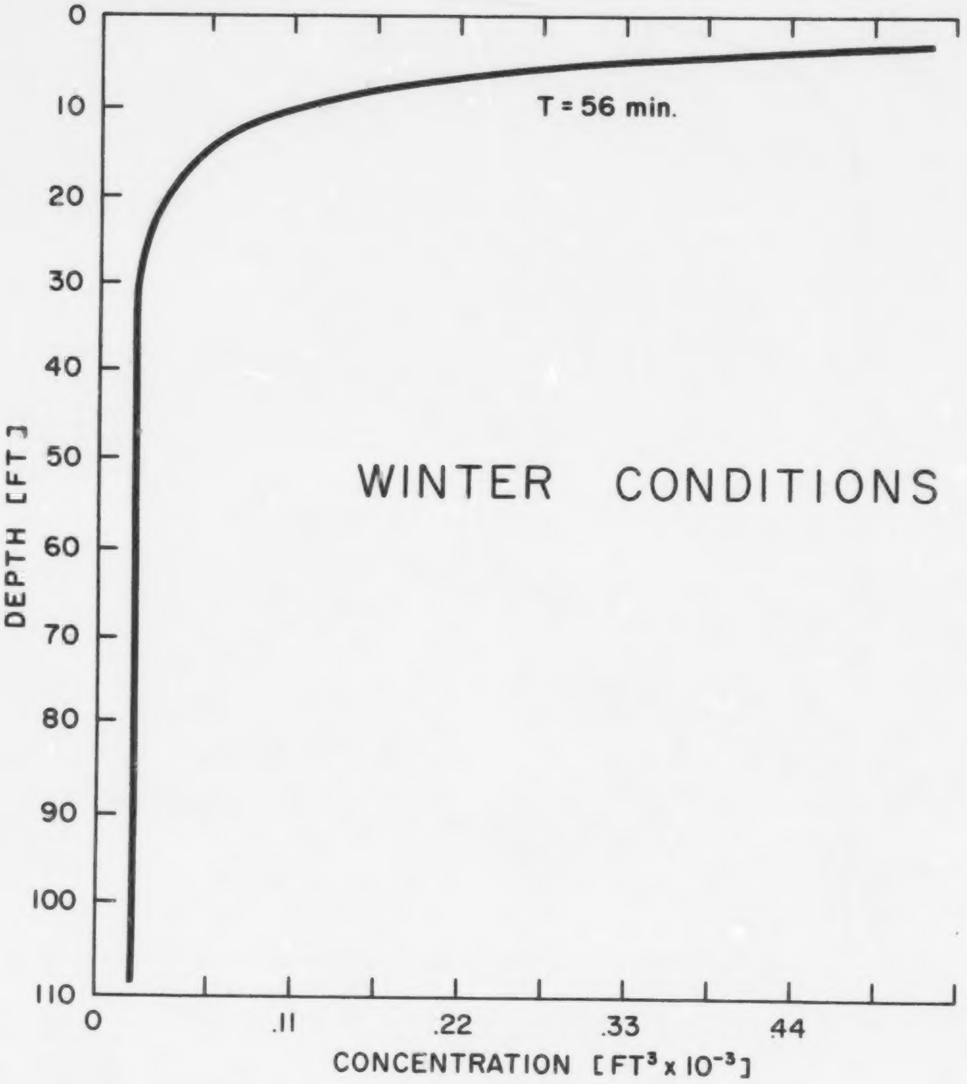

Fig. 10 Vertical Distribution of Relative Wastes Concentrations During Long Term Diffusional Transport; Floatable Solid Phase; Winter Conditions

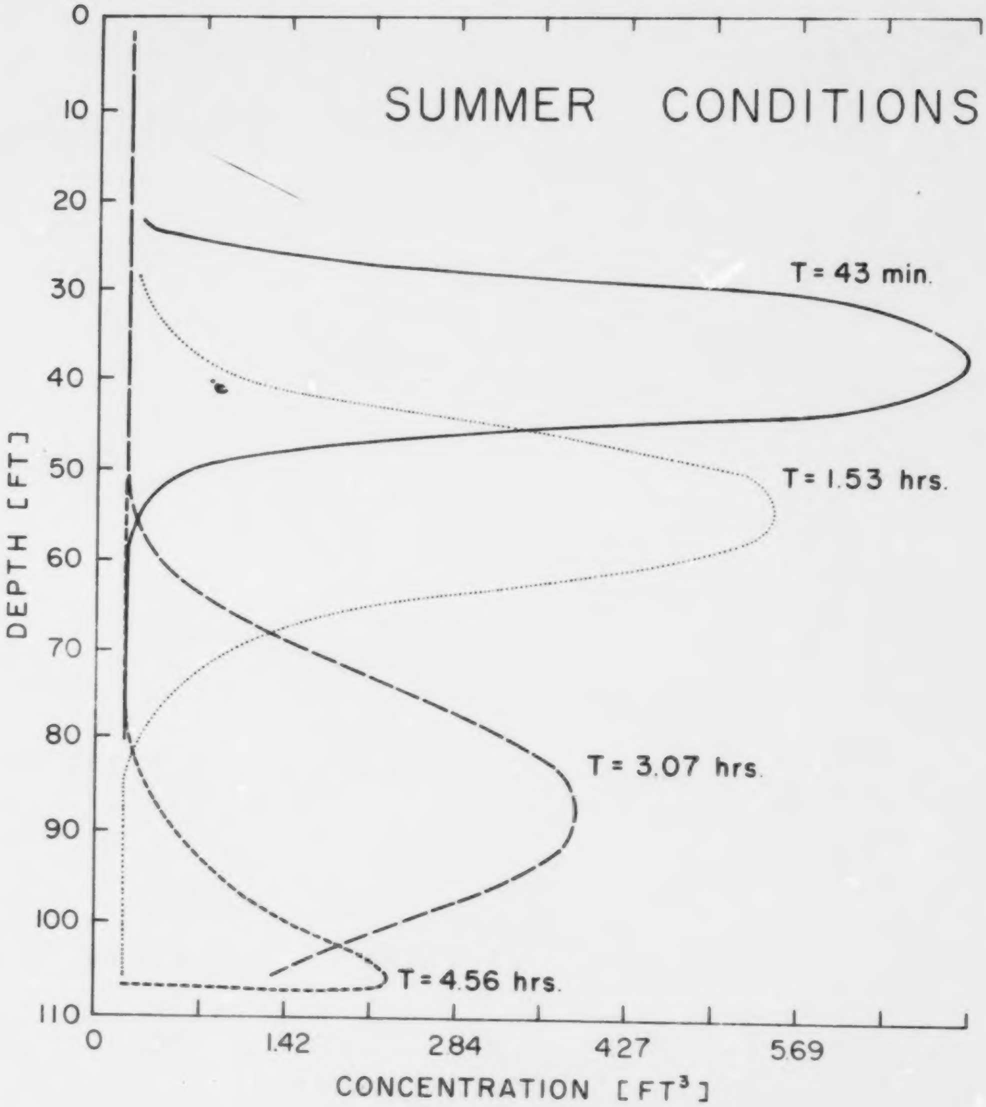

Fig. 11 Vertical Distribution of Relative Waste Concentrations During Long Term Diffusional Transport; Settleable Solid Phase; Summer Conditions

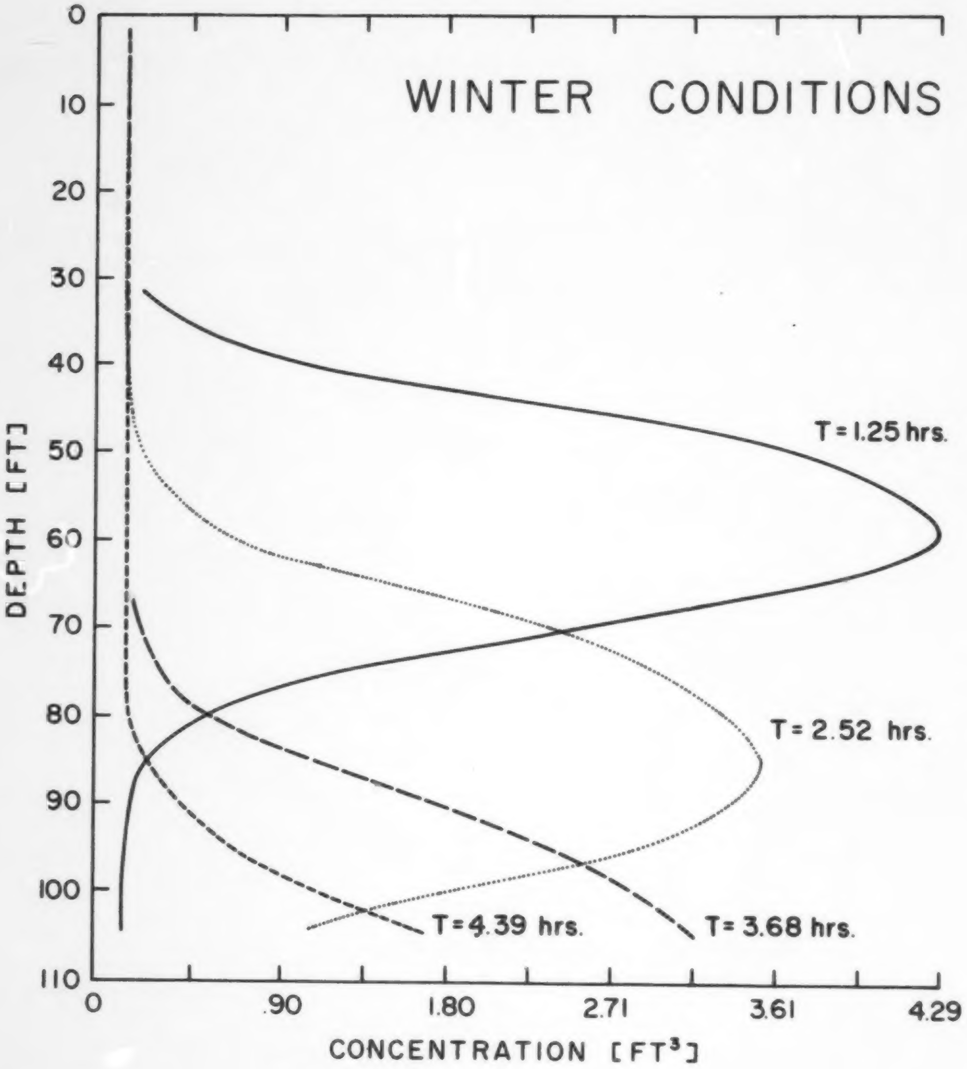

Fig. 12 Vertical Distribution of Relative Waste Concentrations During Long Term Diffusional Transport; Settleable Solid Phase; Winter Conditions

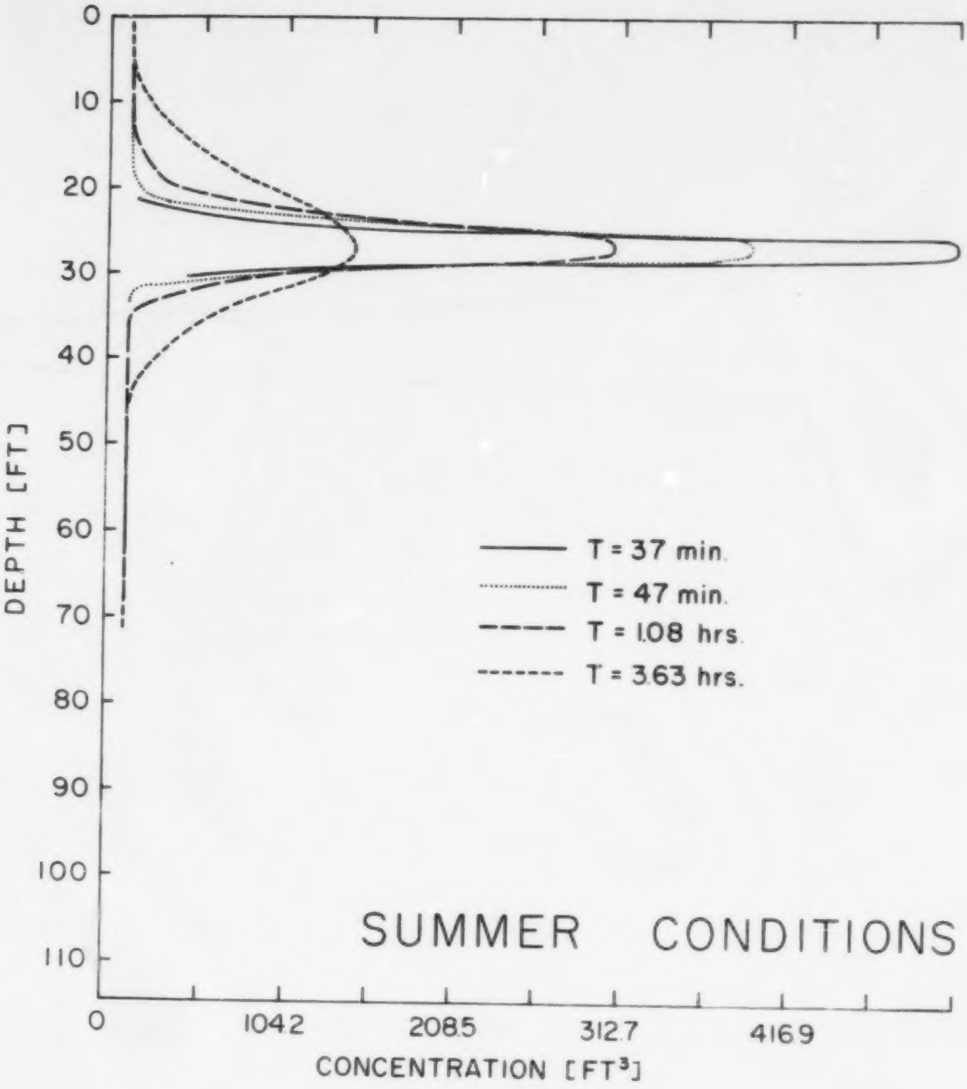

Fig. 13 Vertical Distribution of Relative Waste Concentrations During Long Term Diffusion and Transport; Fluid Phase; Summer Conditions

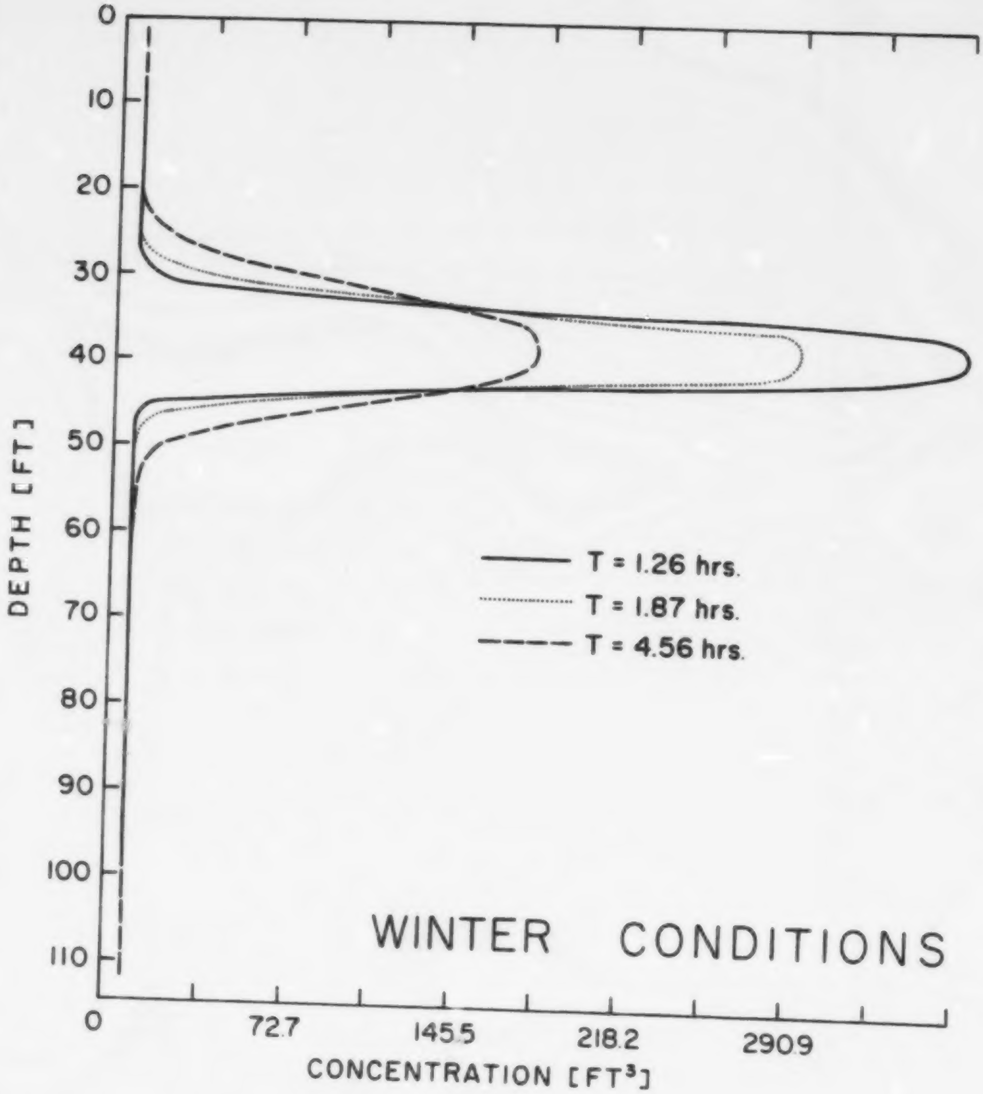

Fig. 14 Vertical Distribution of Relative Waste Concentrations During Long Term Diffusion and Transport; Fluid Phase; Winter Conditions

Figure 7, shows the fluid phase to remain near the surface; the floatables to be quickly deposited on the surface, and the settleable solid phase to be transported several miles from the discharge point. Figure 8, for winter conditions reveals the deeper penetration, the slower release of floatables and the more rapid deposition of the settleable solids. Figures 9-14 reproduced from the computer plots show separate distributions at the time when the maximum relative concentration of each waste has been reduced to prescribed percentages of the initial maximum at the start of this transport phase. In these examples the percentages were 99, 75, 50 and 25. It should be noted that the concentrations vary significantly from graph to graph while appearing to be similar, due to change of scale. To relate these relative concentrations, (cubic feet of waste in a horizontal plane of unit thickness) to actual dilutions, a plot, such as Figure 5 or the tabular output must be compared at similar times.

Depositional Characteristics

The incremental accumulation of material on the boundaries is shown in Figures 15 and 16. These distributions are assumed to be Gaussian and are presented in terms of the percent deposited where the distributions are shown relative to the discharge point and to the maximum amount of deposition that has occurred at the indicated times. The data points shown on the curves denote the centroid location on the X axis and the percent deposited for that curve only. The thickness of the bottom deposit may be estimated through the use of the following equation:

$$h = W/2\pi(\sigma_X{}^2\ \sigma_Z{}^2 - \sigma_{XZ}{}^2)^{1/2}(1-P) \qquad 1.2$$

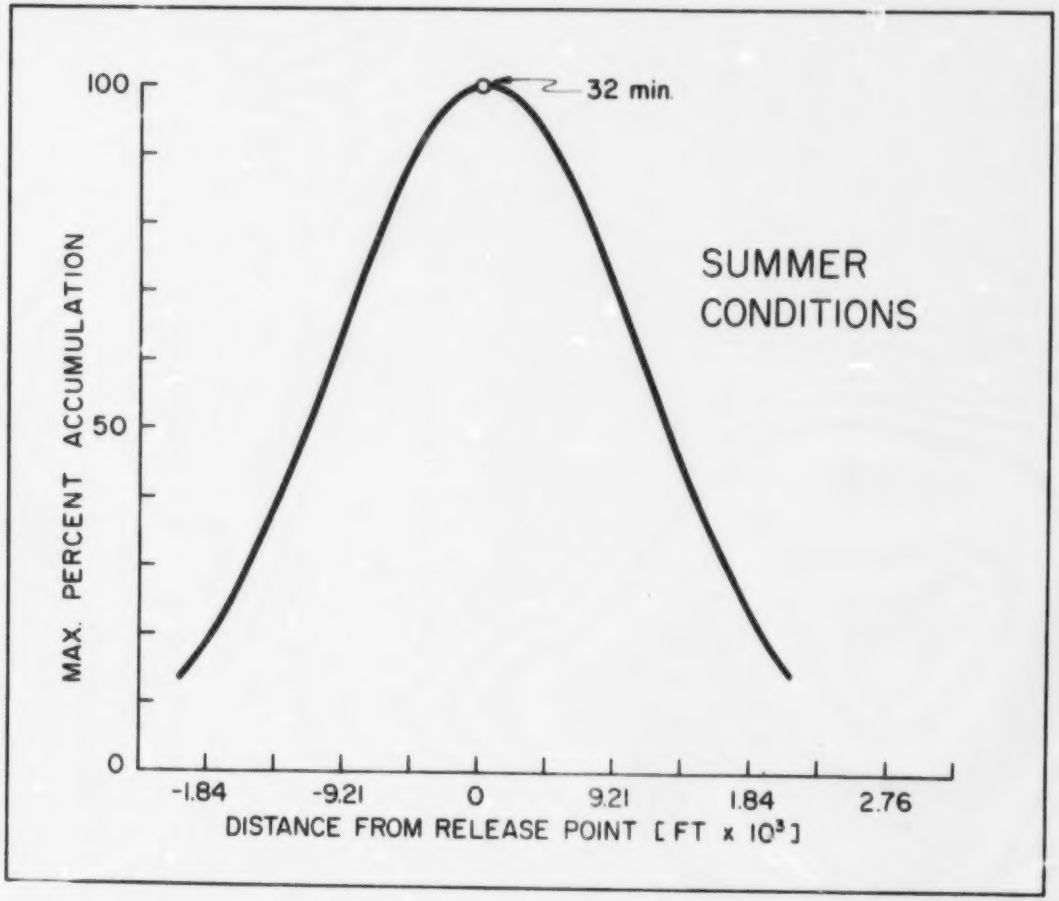

Fig. 15 Distribution of Settled Solids; Summer Conditions

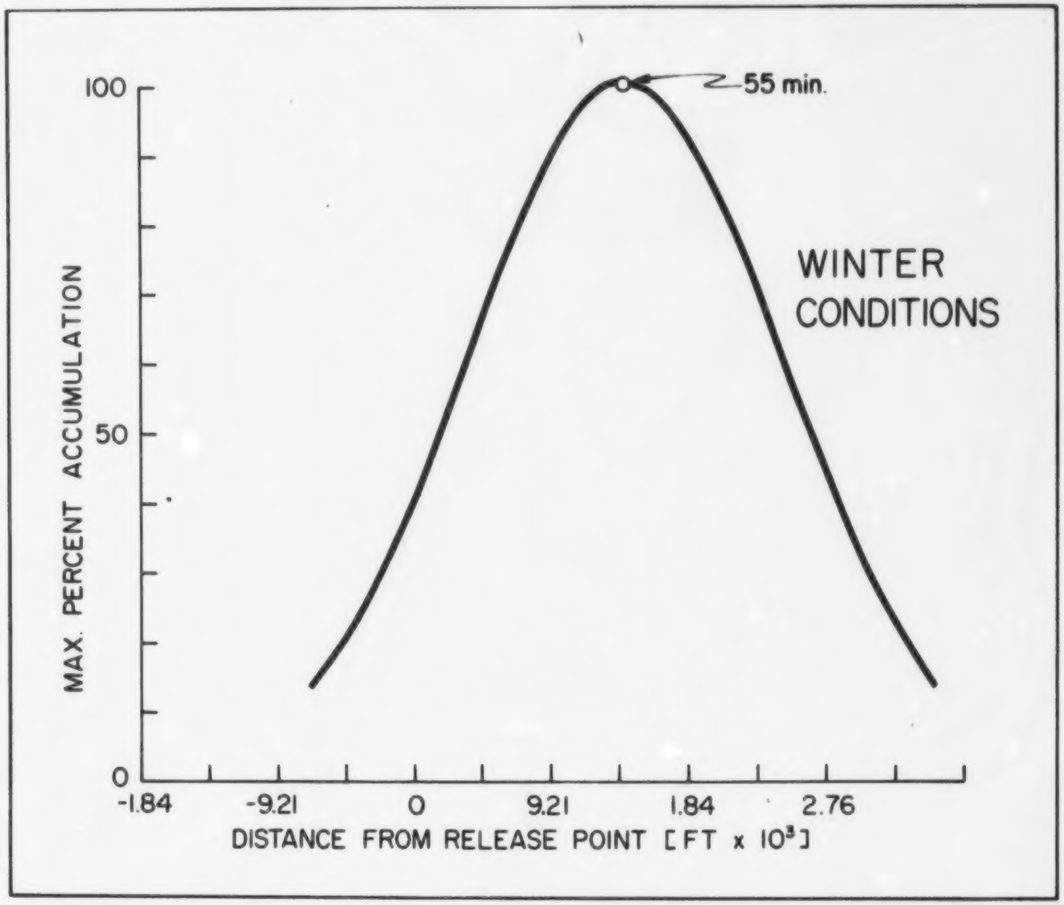

Fig. 16 Distribution of Settled Solids; Winter Conditions

where

h = maximum thickness [ft]
W = total waste discharge [ft^3]
σ_X^2 = X variance of deposited solids [ft^2]
σ_Z^2 = Z variance of deposited solids [ft^2]
σ_{XZ} = covariance of deposited solids [ft^2]
P = porosity

For the examples presented here the maximum thicknesses calculated for porosities of 0.5 and 0.8 are presented in Table 3.

The accumulation of solids on the surface must be expressed differently and, for these examples, the floating solids are presented on a weight per unit area base. The equation for the maximum concentration (D) assuming the solids to be normally distributed is:

$$D = AW\rho_S/2\pi(\sigma_X{}^2\ \sigma_{XZ}{}^2)^{1/2} \qquad 1.3$$

where

D = maximum concentration [gms/m^2]
W = accumulated solids [ft^3]
ρ_S = solid density [gms/cc]
A = constant = 3.02×10^5 [$cc/ft\text{-}m^2$]

Figures 17 and 18 are representative of these distributions and their location relative to the discharge point.

Resuspension of deposited solids was not included in these examples, but is a capability of the model.

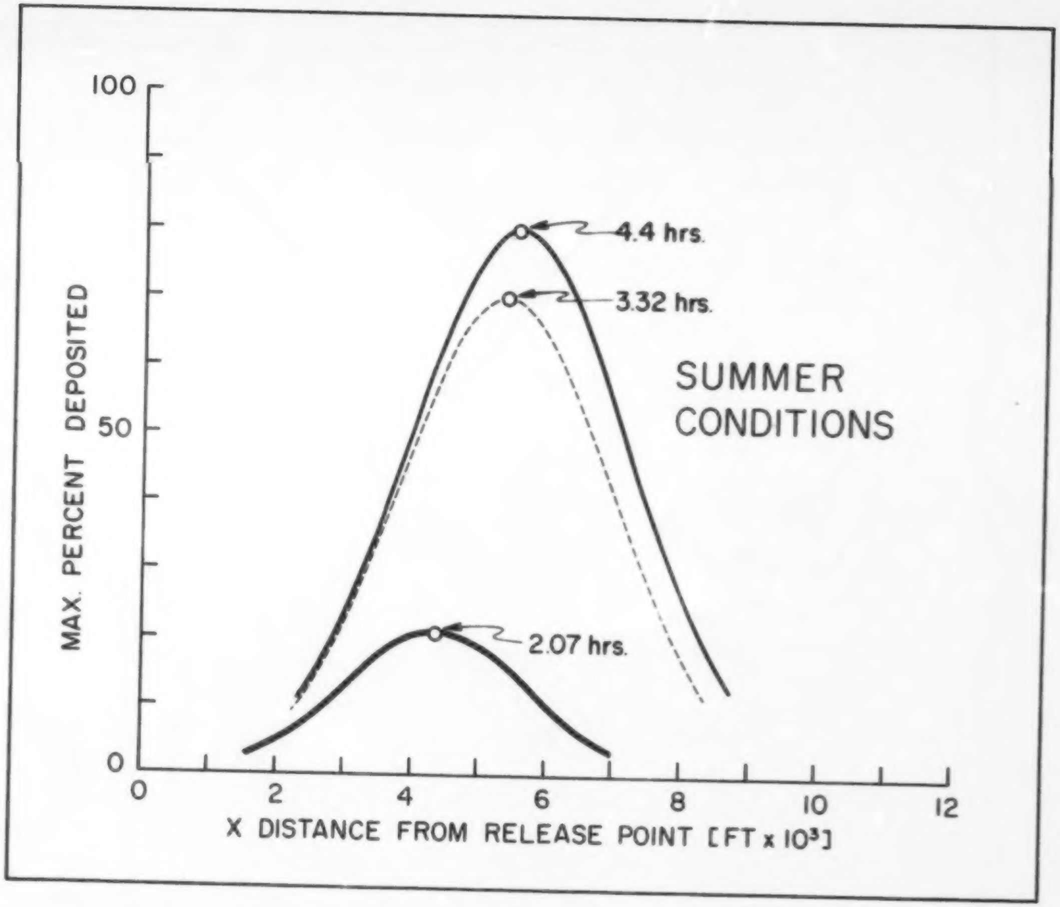

Fig. 17 Distribution of Accumulated Floating Solids; Summer Conditions

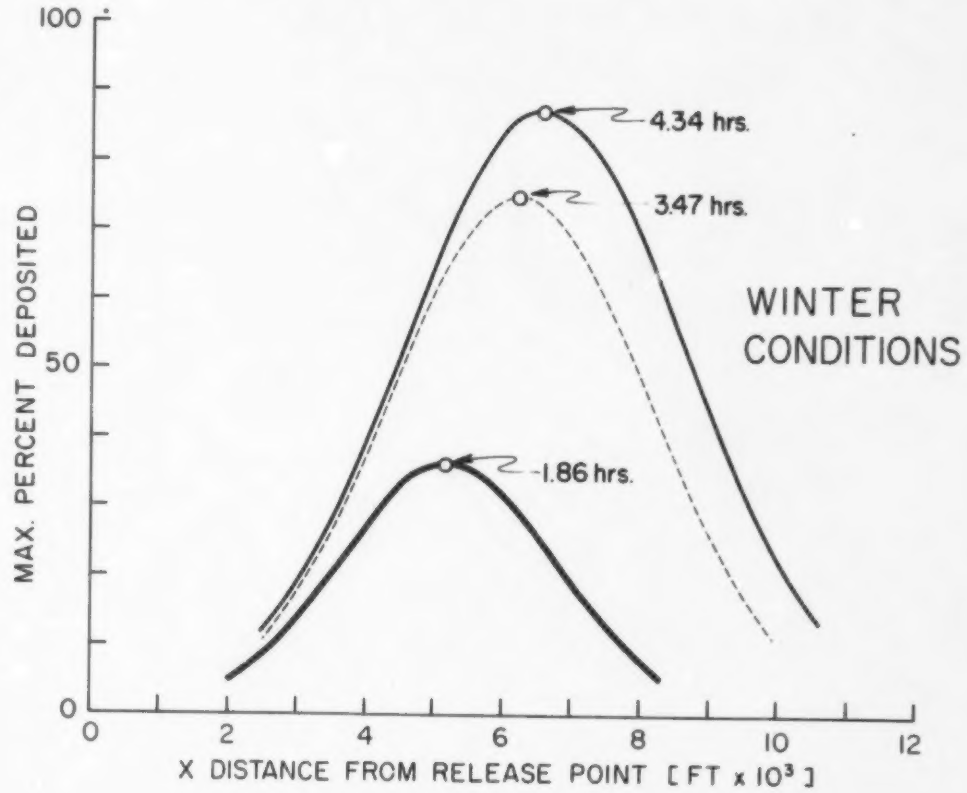

Fig. 18 Distribution of Accumulated Floating Solids; Winter Conditions

Table 3

Calculated Maximum Thickness of Deposited Solids and Maximum Concentration of Floating Solids

Maximum Thickness of Bottom Deposits
Assuming Gaussian Distribution

Season	Time hrs.	% Deposited	Thickness* (microns) p = .5	Thickness* (microns) p = .8
Summer	4.4	80	11.7	29.2
Winter	4.3	86	10.9	27.3

Maximum Areal Concentration of Surface Accumulation
Assuming Gaussian Distribution

Season	Time hrs.	% Accumulated	Concentration* (gms/m^2)
Summer	< 1.0	99+	5.49
Winter	< 1.0	99+	3.79

*Calculations based on single port discharge only.

INTERPRETATION OF DATA

The model output has been used to demonstrate the penetration of the waste cloud, the dilution of each waste phase, the location of maximum waste concentrations as functions of time and space and the areal extent and distributions of the solids on the boundaries. This information can be related to the Federal Regulations (2) controlling release zones, mixing volumes and limiting permissible concentrations.

The release zone is defined as,

> "The area swept out by the locus of points constantly 100 metres from the perimeter of the conveyance engaged in dumping activities, beginning at the first moment in which dumping is scheduled to occur and ending at the last moment in which dumping is scheduled to occur."

This zone is interrelated to a mixing zone by the following:

> "The mixing zone is the region into which a waste is initially dumped or otherwise discharged, and into which the waste will mix to a relatively uniform concentration within four hours after dumping. It is required that the concentration of all waste materials or trace contaiminants be at, or below, the limiting permissible concentrations at the boundaries of the mixing zone at all times and within the mixing zone four hours after discharge. The actual configuration of a mixing zone will depend upon vessel speed, method of disposal, type of waste, and ocean current and wave conditions. For the purposes of these regulations a volume equivalent to that of a mixing zone is the column of water immediately contiguous to the release zone, beginning at the surface of the water and ending at the ocean floor, the thermocline or halocline, if one exists, or 20 metres, whichever is the shortest distance."

The limiting permissible concentration is:

> "(a) That concentration of a waste material or chemical constituent in the receiving water which, after reasonable allowance for initial mixing in the mixing zone, will not exceed 0.01 of a concentration shown to be toxic to appropriate sensitive marine organisms in a bioassay carried out in accordance with approved EPA procedures; or (b) 0.01 of a concentration of a waste material or chemical constituent otherwise shown to be detrimental to the marine environment."

The depth criteria are explicit and for the summer conditions, are descriptive of a thermocline at 10 metres and for the winter conditions equal to the 20 metre maximum. The settleable solids present in sewage sludge penetrate the entire water column and violate by presence, if not by concentrations, these limits. The relative toxicity of the various waste phases must be known before their true importance can be assessed; however, the fluid phase, in these examples, makes up 95 percent of the waste bulk and while not known to be the more important will be examined in that context.

The fluid phases, for both summer and winter conditions as shown in Figures 7 and 8, and 13 and 14, have centroids located above their respective mixing zone depth limits for the entire 4 hour simulation period.

When it comes to examining the lateral extent and movement of the waste for mixing zone compliance, some interpretation is necessary. The more common interpretation assumes the mixing zone to be fixed to the surface and to have the same surface area dimensions as the release zone. The depth is that prescribed by the regulation and is equal to or less than 20 metres. This defined volume is then allowed to be convected. If the movement is identical to that of the time variable maximum waste concentration the problem can be evaluated. This is a relatively simple procedure in the simulated model situation as demonstrated by Figure 19 which shows the relative position and size of the waste cloud and the mixing zone at selected times. The waste cloud dimensions are for the depth where the waste concentration is maximum and do not necessarily correspond to the largest projected surface area. Figure 19 can also be used to visualize the situation where the mixing zone is allowed to move with surface currents, as for example denoted by the movement of a surface drogue. In these examples, the directional current shear with depth would be neglected and movement would be along the X axis-to the benefit of the dischargee, as no appreciable waste concentrations would exist and compliance would be erroneously indicated.

Mathematically, there is another approach which can be used in concert with the computer simulations. A statistical measure of the waste cloud's horizontal size is calculated during the long term transport mode and defined as the geometric mean of calculated standard deviations by the following relationship:

$$\text{Size} = [4\sigma_X \times 4\sigma_z]^{1/2} \qquad 1.4$$

where

$4\sigma_x$ = statistical length of the waste concentration along the X axis [ft]

$4\sigma_Z$ = statistical width of the waste concentration along the Z axis [ft]

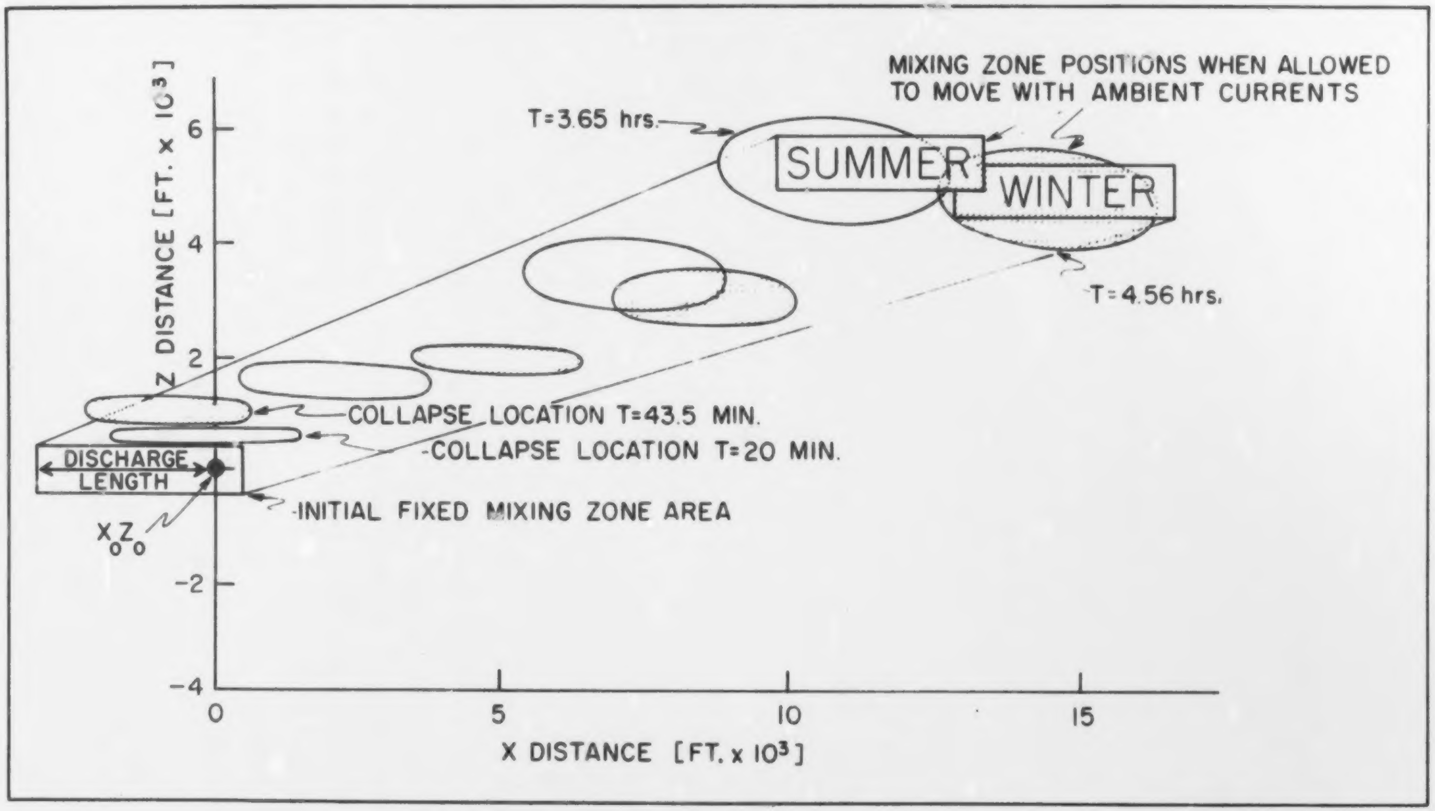

Fig. 19 Plan View Comparing Fluid Phase and Mixing Zone Sizes at Selected Times

A similar equation can be used to define the horizontal size of the mixing volume. This equation is:

$$S = [(200 + L_B + U_B T_d) \times (200 + W_B)]^{1/2} \qquad 1.4$$

where

S = Horizontal release zone [metres]
L_B = Barge length [metres]
U_B = Barge Speed [metres/sec]
T_d = Discharge time [sec]
W_B = Barge width [metres]

The values necessary for this calculation are provided in Table 1 and predict a size for the mixing zone of 506.5 metres (~1660 ft). The computer plots for the size of the waste's fluid phase, Figures 20 and 21, show that the mixing zone size is not approached within the first three hours after discharge. The time series plots represented in these figures are for times where the concentration has been reduced to 99, 75, 50 and 25 percent of the maximum at the beginning of this transport mode; thus, the size can be evaluated directly and the dilution found by referring to Figures 5 and 6 with proper corrections to account for multi-port discharges. Implicit in this approach is an assumption that the movement of both the mixing zone and waste phase of interest are identical. However, when a waste is multi-phased, as in these examples, a more rigorous evaluation of settleable solids and floatables should be undertaken. It may be necessary to explore other interpretations of the mixing zone concept and to include limits for accumulated and deposited materials on boundaries.

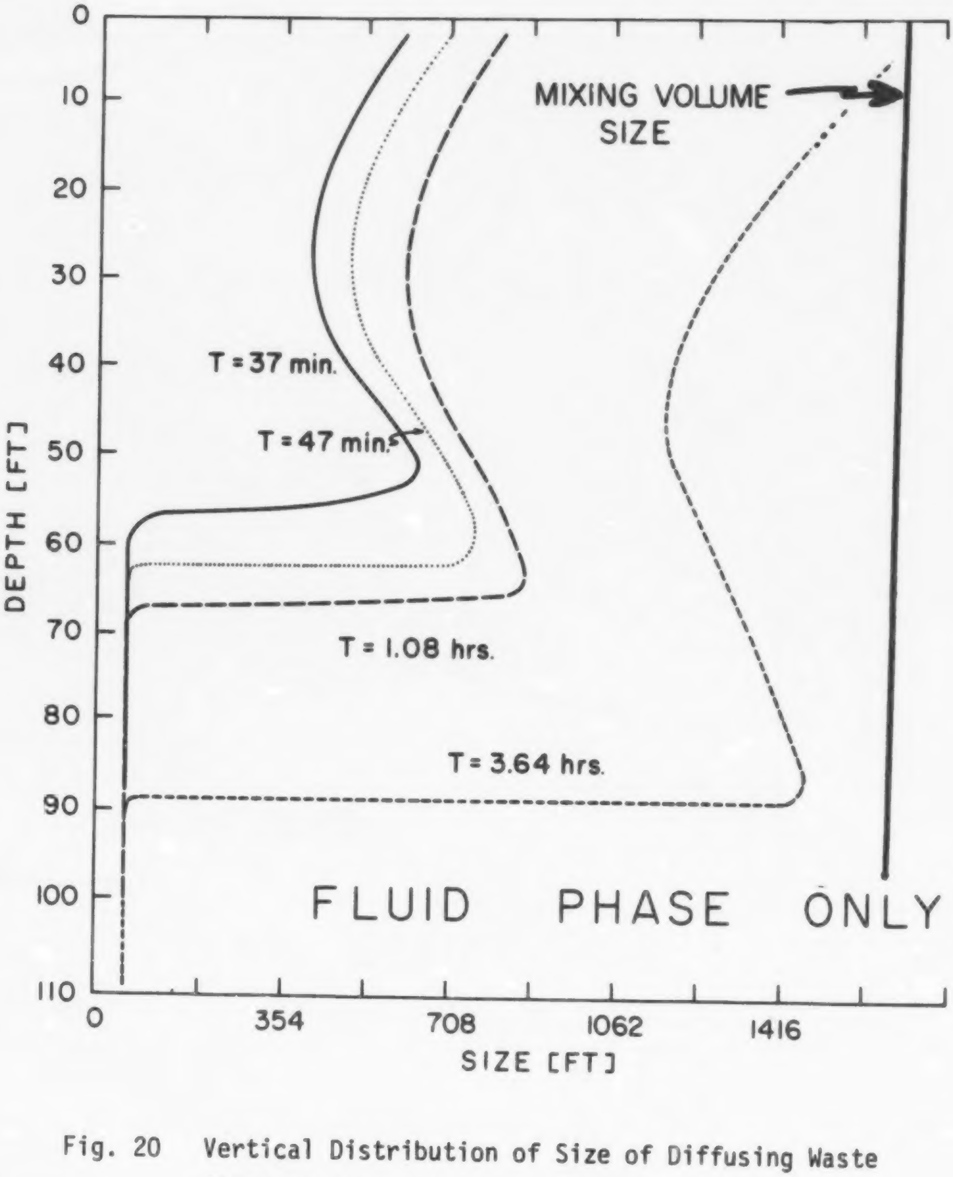

Fig. 20 Vertical Distribution of Size of Diffusing Waste Cloud at Selected Times; Summer Conditions

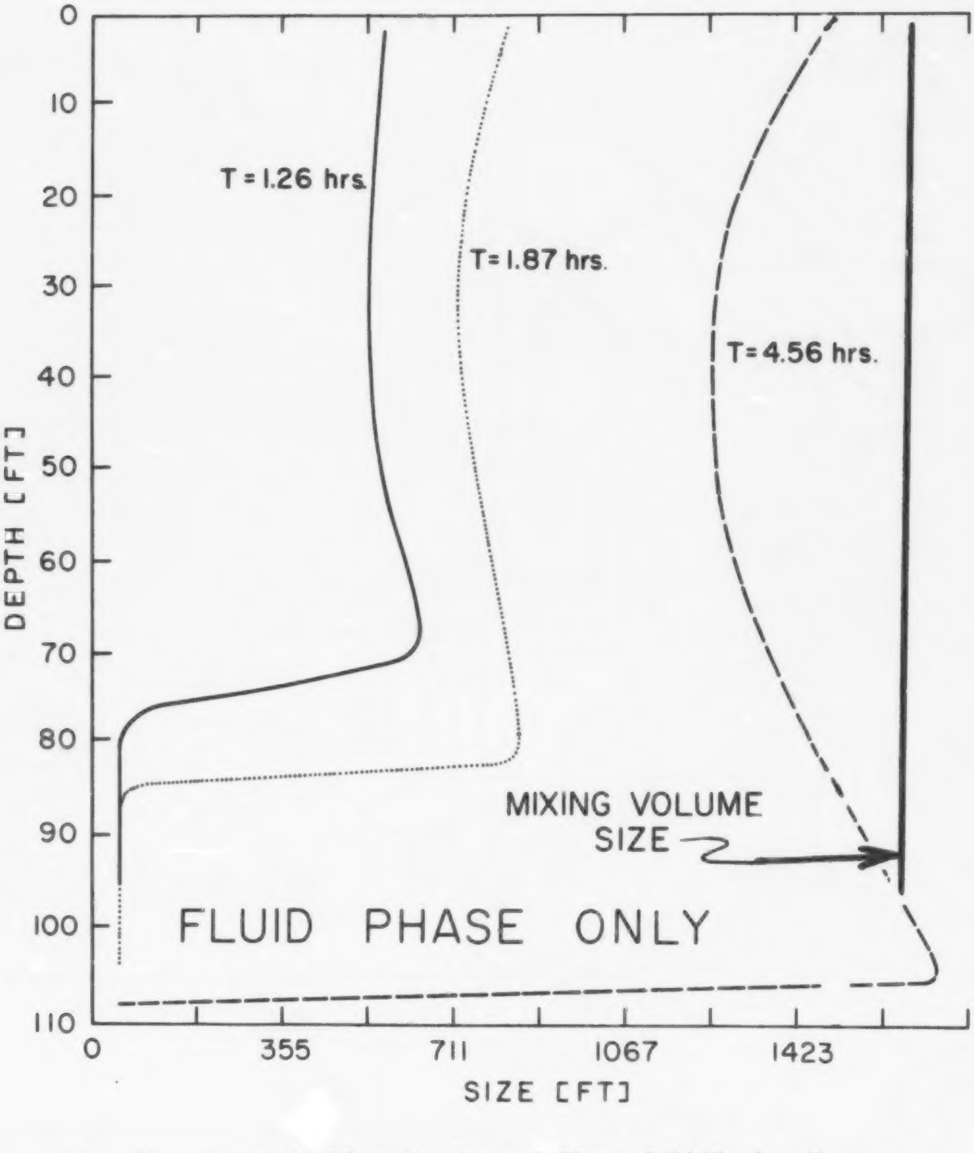

Fig. 21 Vertical Distribution of Size of Diffusing Waste Cloud at Selected Times; Winter Conditions

Limiting Permissible Concentrations

The regulations (2) prohibit discharge of several materials such as mercury and cadmium when present as other than trace contaminants. For sewage sludge, dredge spoils, and the wastes of industries not using or producing these constitutents, their presence is termed a "trace contaminant" and they may be discharged under a special permit when the limits shown in Table 4 are not exceeded.

Table 4

Maximum Allowable Concentrations
for Mercury and Cadmium (2)

Contaminant	Max. Allowable Concentration	
	Solid Phase mg/kg	Liquid Phase mg/kg
Mercury	0.75	1.5
Cadmium	0.60	3.0

Limiting permissible concentrations are also defined in relation to bioassay results and mixing zone boundaries. To examine compliance with the regulations, the model results can be used in the following manner:

The maximum concentration centroid location for the waste phase of interest defines a "worst case" situation. The vertical concentration distribution can be used to obtain estimates of the concentration at a given horizontal boundary. When compliance to the vertical mixing zone boundaries is sought, the concentration and size of the diffusing waste cloud can be obtained for a given depth. The distribution within the cloud for this model is taken to be normally distributed and thus from the model generated standard deviations, concentrations, and the known mathematics of the distribution the concentration at any point can determined. Standard tables (4) are available for the normal distribution and can be used for such evaluations. The height of the distribution is defined in terms of the maximum concentration and the width in terms of the centroid location and the standard deviation along the appropriate horizontal axis. The relationship of these parameters is provided in Table 5.

Table 5
Concentration - Standard Deviation
Relationship for a Normally
Distributed Waste (5)

Fraction of Max. Concentration ($\sigma_x = 0$)	Number of Standard Deviations from Centroid
1.0	0
0.9	0.459
0.8	0.669
0.7	0.844
0.6	1.010
0.5	1.178
0.4	1.353
0.3	1.551
0.2	1.739
0.1	2.145
0.05	2.458
0.02	2.798
0.01	3.034

For the examples presented in Figure 19, where the position of the maximum waste concentration in the fluid phase was compared to the mixing zone position, such estimates have been made. However, for simplicity, the centroid location of the waste cloud was assumed to be at the center of the mixing zone surface area. The boundaries of interest are then equal to one half the X, Z dimensions of the release zone and can be calculated from the data provided in Table 1. The distances obtained can then be expressed in terms of the calculated standard deviations and used with Table 5 to obtain the concentration at the boundary. Table 6 has been prepared to show the results for the two times shown in Figure 19.

Table 6

Calculated Mixing Zone Boundary Concentration
for a Single Port Discharge at
Selected Times (fluid phase only)

	Summer	Winter
time [hrs]	3.65	4.56
σ_X [ft]	1073	1127
σ_Z [ft]	462	628
Mixing Zone X Boundary [ft]	1928	1928
Mixing Zone Z Boundary [ft]	358	358
Max. Concentration	4.6×10^{-5}	3.1×10^{-5}
X Boundary Conc.	9.1×10^{-7}	6.1×10^{-7}
Z Boundary Conc.	3.0×10^{-5}	2.6×10^{-5}

From the data presented the boundary value concentrations are shown to be low with dilutions ranging between 35×10^3 and 1.5×10^6. These values are for single port discharges and could be decreased by as much as three orders of magnitude to adjust for multi-port discharges and the 0.01 factor applicable to demonstrated toxic levels. While a procedure exists - as before - its usefulness depends on the analytical information available as to the toxicant-waste phase relationships and the possible transformations chemical or otherwise that occur subsequent to release.

No specific criteria seem to exist, at present, to allow direct comparisions of model predicted deposited solids to maximum concentrations allowable on either the surface or the seabed.

SUMMARY AND CONCLUSIONS

A mathematical model capable of simulating the barged release of sewage sludge to coastal waters was demonstrated. The results were evaluated in terms of predictive capabilities and applications to the general problem of permit evaluation, site monitoring surveys, and compliance with regulatory criteria. This model is predicted to be of significant value to those with the responsibility to safeguard the marine environment as there is no other procedure or model available to the regulatory programs which provides this degree of analysis (6).

RECOMMENDATIONS

Scientist and engineers specifically involved in ocean dumping studies, permit evaluation, and field monitoring activities are urged to use the model to demonstrate for themselves its usefulness in helping them to meet their specific objectives. The staff of EPA's Coastal Pollution Branch is willing to provide assistance; however, before this is done the requirements of the model should be fully understood. As any simulation is extremely dependent on the data chosen to describe the system, it is important to acquire sufficient data to accurately characterize the waste, the barge, and the environment.

Work is ongoing at many locations to improve the model and to evaluate it under field conditions. The US Army Corps of Engineers, Waterways Experiment Station has funded a program to refine the model and make it more responsive to dredging operations in estuaries. Use by two non-government contractors working on the problem of open water spoil disposal is currently in the planning phase.

It will be found by those who attempt to utilize the model that existing field and analytical data will not always be sufficient to meet the input requirements, and some additional work may be required. The benefits, however, will far exceed the costs. The model is destined to become an intergral part of the ocean dumping evaluation process and those undertaking independent research and refinements are urged to make the results available to prevent duplication of effort and unnecessary delays in the realization of the model's true potential.

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